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Extent and Magnitude of Recent Changes in Forest Condition and the Role of Air Pollution and Non-Air Pollution Factors

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The Forest Response Program(FRP) is a research initiative under Task Group V of the National Acid Precipitation Assessment Program (NAPAP). In response to reports of forest decline, the FRP was established to estimate actual and potential effects of acid deposition and its associated pollutants on trees, forests and forest ecosystems in the regions of the United States.

The purpose of this document is to synthesize research on two scientific questions,

1) Are changes in forest condition greater than can be attributed to natural variability? and,
2) What spatial patterns exist in forest condition and how do they relate to spatial patterns of exposure?

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1 EXECUTIVE SUMMARY

The purpose of this document, Major Program Output #1&2, is to summarize research addressing two scientific questions:

1) Are changes in forest condition greater than can be attributed to natural variability?

2) What spatial patterns exist in forest condition and how do they relate to spatial patterns of pollutant exposure?

Emphasis has been placed on summarizing research conducted within the Forest Response Program (FRP).

1.1 Changes in Forest Condition

To answer the first question, both measures of forest condition and sources of natural variability in forest condition must be identified. Measures of forest condition used by the FRP are tree and stand growth, tree mortality, tree crown condition, and changes in soil nutrient levels. Sources of natural variability in forest condition are tree age, stand density, site quality, climate, insects, pathogens, fire, and genotype. Management practices are another source of variability associated with forest condition. Individual FRP studies generally evaluated one or more measures of forest condition and accounted for some but not all of the sources of variability.

Based on the results of the research discussed in this document, two regions can be identified that show changes in forest condition. First, the incidence of ponderosa pine foliar injury in the southern Sierra Nevada of California is greater than what would be expected from natural sources of variability. Second, in the Northeast, high-elevation red spruce showed increased mortality in the 1960s. Whether or not the change in red spruce mortality is outside the range of natural variability is not known, since a red spruce mortality episode of similar magnitude occurred in the late 1800s. In addition, these high-elevation sites are by their nature under high stress due to factors such as wind exposure, climate, tree competition, and shallow soils.

1.2 Spatial Patterns in Forest Condition and Pollutant Exposure

The second scientific question has two components. First, are there spatial patterns in forest condition either among regions or at different elevations in one region? To answer this question, the sources of natural variability must have been accounted for at each site before comparisons can be made among sites. If there are spatial trends, the question then asks whether these patterns are associated with similar patterns in atmospheric deposition. To determine this answer, both forest condition and deposition levels must be known at coincident locations. However, deposition monitoring typically is not conducted at the same sites at which forest condition is assessed. Projects estimate rather than actually monitor pollutant levels at forest plots, and the variability of these estimates is either quite large or unknown. Thus, only forest plots that are a great distance apart are known to have significantly different levels of pollutant deposition. Identification of spatial patterns in deposition over time is further limited because monitoring data extend back only to the late 1970s, and patterns vary greatly from year to year. Thus, there has been limited success in answering the second scientific question.

Based on the research presented in this document, spatial patterns in altered forest condition appear in two areas. First, in the southern Sierra Nevada of California, injury to ponderosa pine needles is consistent with symptoms typical of ozone exposure, and the injury is found in areas experiencing elevated levels of ozone. Second, in the Northeast, the percentage of red spruce standing dead basal area increases with increasing elevation in the Adirondack and Green Mountains. Although it is not known if the proportion of standing dead red spruce is outside the range of natural variability, the increases are consistent with increasing levels of wet deposition.

1.3 Summary of Principal Findings

The research used to answer the two scientific questions is summarized in this document by forest type and region.

1.3.1 Northern Red Spruce

Prior to and during the initiation of the FRP, studies of red spruce across the Northeast indicated that increased mortality at high elevations and radial growth reductions at both high and low elevations had occurred in the mid-1960s. These reports led researchers to believe that there had been a large-scale synchronous decline in red spruce condition. However, before this change in forest condition can be related to deposition levels, sources of natural variability must first be considered.

Mortality

The number of standing dead red spruce at high elevations increased between the mid-1960s and mid-1980s in the Adirondack, Green, and White Mountains. Mortality episodes appear to be related to frequent winter injury events that started in the late 1950s and early 1960s. At the highest elevations (above 1200 m), 30% to 60% of red spruce basal area was dead. The Adirondack and Green Mountains showed an increased percentage of standing dead red spruce with increasing elevation. The greatest mortality has occurred above 1000 m, an elevation that may represent an extreme in the range of red spruce. Thus, an environment that may be considered marginal for red spruce may become intolerable for the species if additional stresses are imposed, such as unfavorable climate and/or acidic deposition. At low elevations (below 800 m), mortality is considered to be within expected ranges.

Growth

Half of the red spruce population in the northeastern United States is in northern Maine, and the remaining population is in southern Maine, New Hampshire, Vermont, and New York. Red spruce radial growth data indicate a widespread pre-1950 radial growth increase and a subsequent post-1960 growth decrease at low and high elevations throughout the northeastern United States. The percentage of sites in the region that show the 1960s growth reduction is unknown because data were aggregated over sites and because studies were not specifically designed to determine this percentage. One study restricted to northern Maine suggests that approximately 54% of low-elevation sites sampled show a pre-1950s increase followed by a post-1960s decrease.

Several natural factors have been suggested to explain this change in radial growth throughout the region. First, extremes in climate (warm summers and cold winters prior to the year of growth) may have been an initiating factor. Both old- and second-growth stands at high and low elevations show this regional climate variation. Research at high elevations suggests that closely spaced years of winter injury in the late 1950s and early 1960s which resulted in loss of current-year needles could be the cause of radial growth reductions and a breakdown of the high-frequency growth-climate relationship since the early 1960s. However, because a number of sites do not show a reduction in growth since 1960, climate does not appear to be responsible for a synchronous growth decline in stands throughout the Northeast. Other research suggests that stand history events could explain the apparently synchronous changes in radial growth at both high and low elevations, independent of a regional climate signal found in all age classes. To test these various hypotheses, future work should correlate changes of radial increment with winter injury and stand history for individual trees at a number of sites.

Crown Condition

Surveys of red spruce crown condition at a range of elevations in New York, Vermont, New Hampshire, Massachusetts, and West Virginia indicate that crown conditions have deteriorated since 1985. There were no changes in mortality over the same period. At Mt. Abraham, Vermont, crown deterioration was associated with slower radial growth since the early 1960s, and trees with healthy crowns had greater radial increment than trees with less healthy crowns.

1.3.2 Southern Red Spruce

In the southern Appalachians, concern over red spruce and Fraser fir exists because of concentrated areas of standing dead trees and possible growth declines of surviving trees.

Mortality

In the southern Appalachians, Fraser fir mortality from the balsam woolly adelgid has increased since the early 1960s. In spruce-fir forests of this region, 24% of the total area has more than 70% of dominant and codominant trees standing dead, and 70% of the total area has less than 30% standing dead trees. Areas with the greatest standing dead were dominated by Fraser fir that had been killed by the adelgid. Red spruce standing dead typically averaged less than 10% throughout the region, and recent mortality events have been associated with ice storms. In West Virginia, only 7% of the total land area of spruce type has greater than 10% mortality. Mortality greater than 30% occurred in 1.5% of the area.

Growth

Since the mid-1960s, radial increment of red spruce at the highest elevations has reduced slightly. Climate, air pollutants, and the balsam woolly adelgid have been suggested as contributors. The effects of past disturbances and the resulting influences of stand dynamics have not been investigated. Radial growth and pollutant exposure have not been correlated because deposition estimates and tree-ring data were obtained from different locations.

Crown Condition

Red spruce crown condition has generally declined from 1985 to 1988. Two ice storms in the winter of 1986-1987 resulted in crown damage to 28% of surviving trees at the highest elevations. During the same period at lower elevations, the number of trees with more than 90% of their crown intact decreased by 20%. In a survey in West Virginia, 40% of the red spruce trees at elevations above 1000 m had live-crown ratios less than 40%, with most of the low crown ratios associated with natural crown suppression. It thus appears that trees with low crown ratios are to be expected.

1.3.3 Southern Pines

In the early 1980s, preliminary analyses of forest inventory data indicated that radial growth of southern pines may be declining. The possibility that these reductions may be associated with air pollution led to further analysis of survey data. Although information is available on southern pine crown condition or mortality, no analyses of these data have been completed.

Growth

Comparisons of the fifth USDA Forest Service Forest Inventory and Analysis (FIA) inventory with the third and fourth inventories indicate that average annual radial increment of southern pine species in natural stands declined by 30% to 50% in the period between the third (1957-1966) and fifth (1977-1985) surveys. Although most of the growth reduction has occurred on nonindustrial private forest land, some reductions have also been identified on public and industrial land. There is some indication that loblolly pine growth rates may be cyclic. Research on growth and yield plots at the Duke Forests indicate that after adjusting for stand density, data from permanent plots that were established in the early 1930s show less growth increment during 1965-1984 than from 1934-1965, and, since 1984, growth rates have increased. Comparisons with climate data suggest no obvious correlation.

Individual tree growth reductions have not been firmly established, nor have any studies clearly identified a decrease in productivity per unit area. The radial growth reduction of diameter classes of trees is expected given that the population of southern pine trees has changed over the last three decades. The trees from early decades were from lower density younger stands, but the most recently sampled populations represent higher density older stands. The populations of trees differ substantially between the time periods studied with respect to the important concomitant variables. The data ranges do not or only slightly overlap.

1.3.4 Eastern Hardwoods

Eastern hardwood forests are subjected to varying levels of pollutant deposition. Because strong spatial gradients in pollutant deposition exist in this area, research was conducted in these forests to investigate the possible influences of pollutants on forest health and productivity.

Mortality

In a historical review of hardwood declines, many species were reported to have experienced mortality events during the last century due to weather, silvicultural practices, pathogens, or insects. There was no evidence of a consistent association between patterns of mortality and regional atmospheric deposition. However, preliminary results from the Ohio Valley indicate that black oak mortality rates have recently doubled from 0.7% to 1.2% per year and are higher in the eastern region of the study, where sulfate deposition is the greatest.

Growth

Research on eastern hardwoods indicates no widespread changes in tree growth. However, some weak associations exist between tree diameter growth and wet deposition in the Great Lakes region (Minnesota to Michigan). After adjusting for climate, diameter growth of two hardwoods (sugar maple and aspen) was positively correlated with estimated sulfur deposition, and diameter growth of two conifers (jack pine and red pine) was negatively correlated with sulfur deposition. A jack pine budworm outbreak that increases along the sulfur deposition gradient may be contributing to decreases in jack pine and red pine growth rates. In another study along this gradient, preliminary analyses show that sugar maple growth rates in Michigan are not directly related to deposition levels, and there is no evidence of decreased productivity. A study in Pennsylvania found no relationship between growth and wet deposition. Preliminary results from the Ohio Corridor gradient study indicate that up to 33% of the black oaks and 20% of white oaks on sites with low soil Ca:Al ratios in the B horizon show reductions in growth rates outside recent historic patterns. In Vermont, of 11 hardwood species studied, none show decreases in growth rates for the most recent time period (1973-1983), and eight species show increased growth.

Soils

From northeastern Minnesota to southwestern Michigan, the amount of sulfur and nitrogen cycled in plant litter increased along a gradient of increasing atmospheric deposition of sulfur and nitrogen. Increasing concentrations of sulfur in the forest floor and soil from west to east

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have also been reported. A survey in 1987-1988 in the Ohio Valley shows a general increase in total soil sulfur and soil carbon concentration along the gradient.

1.3.5 Western Conifers

Chronic symptoms of ozone damage to ponderosa and Jeffrey pine in the San Bernardino Mountains of southern California have been documented for over two decades. Ozone concentrations have damaged foliage of trees. Concern that foliar injury may lead to changes in forest growth prompted studies to determine whether there have been recent changes in tree growth. No information is available on western conifer mortality.

Growth

In the southern Sierra Nevada (north of the San Bernardino Mountains), sites exposed to ozone have shown fewer increases in radial growth compared with sites that are exposed to less ozone. Because sites exposed to increased levels of ozone show a reduction in the number of individual trees with growth increases since 1950 along with increased foliar injury, ozone has been suggested as a possible cause. In southern Arizona, there is evidence of some radial growth reductions in ponderosa pine and Douglas-fir. Researchers have suggested that local pollutant deposition may have affected the climate-tree growth relationship, especially in the south. However, sites located near urban areas (and thus with elevated ozone exposure) show no consistent growth reductions. Natural competition among trees may be responsible for some of the growth decreases.

In the Front Range of the Colorado Rocky Mountains, the climate-tree growth relationship has shown no widespread change. Radial growth of Douglas-fir increased in the 1980s and was correlated with increased precipitation. In the Puget Sound area of the Pacific Northwest, radial growth of Douglas-fir increased from 1880 to 1950, remained constant or decreased slightly until 1975, and then increased. The overall increase since 1880 is probably related to a regional temperature increase. No relationships suggest a pollution effect.

Crown Condition

Ozone exposure increases from north to south in the southern Sierra Nevada of California, and crown injury to ponderosa pine increases over a 250-km corridor in the same direction. The greatest foliar injury is in Sequoia-Kings Canyon National Park. In the Colorado Front Range, no visible injury has been found, although ozone exposure was at levels associated with chlorosis of needles in the southern Sierra Nevada. This finding suggests that either the duration of exposure to elevated levels of ozone is less or ponderosa pine in the Front Range are more resistant to ozone injury than are ponderosa pine in California. In Arizona, ozone damage has been reported for the Rincon Mountains, where 23% of the trees showed foliar injury.

1.3.6 Wet Deposition

Total wet deposition due to clouds and precipitation in eastern forests was greatest at high elevations with frequent cloudiness. In the West, wet deposition of pollutants was generally less at locations distant from point sources and cities. For areas below cloud base in the Northeast, upper and lower Midwest, and Southeast, precipitation deposition of hydrogen, sulfate, nitrate, and ammonium ions was relatively high across each region with no true gradient demonstrated. In states near the edge of these regions (Minnesota, Maine, Arkansas, and Florida), deposition via precipitation was lower than in the remaining portions of the region (Michigan, New York, Illinois, and North Carolina).

In general, the highest deposition of ions by precipitation occurred in the Ohio Valley and northeastern states, and there was relatively low deposition in western forests. Chemical deposition from cloud water in high-elevation areas of the Northeast and Southeast appeared comparable to or greater than deposition from precipitation. A more limited data base indicates that the same may be true in the West. Although direct correlation to changes in forest growth and condition have not been possible, these results illustrate potentially increased risks from pollutants at high-elevation sites.

1.3.7 Ozone

Most ozone monitoring has been conducted in or near urban areas. Although ozone from these areas may impact forests because of long-range transport, little is known about actual levels of ozone in forests. Available data indicate that across the United States, areas in southern and central California, the high desert and intermountain region of the West, and the southcentral East and midwestern states were more likely to be exposed to elevated ozone levels than were other areas. In the Northeast, high-elevation forests experienced average ozone concentrations of between 40 to 60 ppb and some very high peak levels. In the southern Appalachians, ozone levels increased with the proximity of the site to urban areas, and median ozone concentrations increased with elevation at remote locations. In midwestern states, forests studied in the southern portion of the region were exposed to very high ozone concentrations. Elsewhere exposures were lower, with few peaks exceeding 120 ppb.

In the West, forests in Arizona and New Mexico experienced hourly ozone concentrations less than 90 ppb, with no distinct spatial pattern in exposure. Forests on western and southern slopes of the San Gabriel and San Bernardino Mountains were exposed to very high hourly ozone concentrations (median concentrations greater than 100 ppb), and direct injury to vegetation has been documented in the San Gabriel and San Bernardino Mountains of California and the Rincon Mountains in Arizona. Forests in the southern Sierra Nevada were exposed to higher concentrations more frequently than forests to the north. On western slopes of the Cascades, south and southeast of Seattle-Tacoma, forests were exposed to higher concentrations than sites elsewhere in the Puget Sound Basin. Finally, forests in the Front Range of Colorado north of Denver were probably exposed to median ozone concentrations of approximately 60 ppb, with peaks up to 130 ppb. For other high-elevation Rocky Mountain sites, exposures of 30 ppb were common.

2 INTRODUCTION

The Forest Response Program (FRP) is a research initiative under Task Group V of the National Acid Precipitation Assessment Program (NAPAP). In response to reports of forest decline, the FRP was established to estimate actual and potential effects of acid deposition and its associated pollutants on trees, forests, and forest ecosystems in regions of the United States (Schroeder and Kiester, 1989). The FRP is jointly administered by the US Environmental Protection Agency (EPA) and the USDA Forest Service (USFS). Operationally, the FRP consists of a National Management Staff, six Research Cooperatives, and three projects (Quality Assurance/Quality Control [QA/QC], Integrated Assessment, and Synthesis and Integration [S&I]).

The objective of the FRP is to address three questions related to environmental policy.

Policy Questions

1) Is there significant forest damage in North America caused by acidic deposition, alone or in combination with other pollutants?

2) By what mechanisms does acidic deposition, alone or in combination with other pollutants, contribute to forest damage in North America?

3) What is the dose-response relationship between acidic deposition, alone or in combination with other pollutants, and forest damage in North America?

The FRP attempts to answer these questions through many diverse projects. Major Program Output (MPO) documents provide program-wide summaries of research addressing the three policy questions.

2.1 Purpose of the Document

The purpose of this document, MPO #1&2, is to synthesize research on two scientific questions that address Policy Question #1.

Scientific Questions

1) Are changes in forest condition greater than can be attributed to natural variability?

2) What spatial patterns exist in forest condition and how do they relate to spatial patterns of pollutant exposure?

To answer the first question, both measures of forest condition and sources of natural variability in forest condition must be identified. Measures of forest condition used by the FRP are tree and stand growth, tree mortality, tree crown condition, and soil nutrient levels. Sources of natural variability in forest condition are tree age, stand density, site quality, climate, insects, pathogens, fire, and genotype. Management practices are another source of variability associated with forest condition. Individual FRP studies generally evaluated one or more measures of forest condition and accounted for some but not all of the sources of variability.

The second scientific question has two components. First, are there spatial patterns in forest condition either among regions or at different elevations in one region? To answer this question, the sources of natural variability must have been accounted for at each site before comparisons can be made among sites. If there are spatial trends, the question then asks whether these patterns are associated with similar patterns in atmospheric deposition. To determine this answer, both forest condition and deposition levels must be known at coincident locations.

In this document we thus report on the extent and magnitude of recent changes in forest condition in the United States, and we discuss whether changes can be explained adequately by natural factors. Data on atmospheric deposition patterns are also presented. Potential mechanisms for changes due to atmospheric pollutants are addressed in MPO #3 (C. Peterson et al., 1989) and MPO #4 (Mattson et al., 1989).

The majority of the research projects cited in this document were funded by the FRP; when appropriate, relevant research conducted outside of the FRP is included to improve interpretations. The fact that these studies contribute evidence to a portion of the scope of the questions, and that methods frequently differ substantially, precludes formal inductive conclusions. However, when consistent results appear for a specific region and species type, deductive inferences can be made.

2.2 Historical Perspective

The recent discussion of acidic deposition as an environmental problem originated in the mid-1950s in Scandinavia (Rossby and Egner, 1955). Monitoring networks were established to study regional patterns of the chemical constituents and acidification of precipitation as well as the possible consequences to ecosystems. Initial research focused on lakes and streams (Bolin et al., 1971; Overrein et al., 1980). However, studies in northeastern United States forests (Likens et al., 1977) and the Solling area of Lower Saxony in West Germany (e.g., Ulrich et al., 1979) showed that the vegetation itself contributes substantially to acid input by filtering air and cloud masses.

Initial indications of a decline in forest health in the Federal Republic of Germany (FRG) were observed in the early 1970s. Chlorosis and crown-thinning of silver fir were reported in so-called "clean air areas" in remote regions of southern Germany. Since this species is sensitive to environmental changes, the symptoms were mainly of local concern, but they did increase awareness of possible changes in forest health elsewhere. Concern increased when, from the mid-1970s to the early 1980s, similar symptoms were observed in widely distributed stands of Norway spruce. Spruce constitutes about 39% of German forests and is the most important commercial species. In the mid-1970s, the decline symptoms in FRG forests became associated with "acid rain." Popular discussions spurred political efforts to intensify environmental protection and increase research initiatives. However, "acid rain" also became a generic term for all ecosystem damage that was in any way attributable to air pollutants (e.g., Brandt, 1987).

Damage to deciduous trees, particularly European beech, has increased in the FRG since about 1984. Because of the decline's rapid development and extent, the situation was popularly known as "Waldsterben" (forest death). However, since many symptoms were assumed to be new phenomena, and to avoid negative connotations, the term "neuartige Waldschaeden" (new-type forest damage) was introduced. The widespread rapid decline in the FRG led to a nationwide census of injury in 1982, and subsequent inventories have been made yearly (e.g., Ballach and Brandt, 1983,1985; Blank, 1985; Blank and Roberts, 1989). Injury classes in these inventories are based primarily on degree of foliage loss and discoloration. Early classifications of trees with greater than 10% foliage loss as damaged were criticized as not accounting for natural variability in crown condition of healthy trees and ecotypes in different regions. Damage classes were redefined, and foliage loss up to 25% is now considered an early warning stage. This redefinition means that less than 20% of FRG forests are classified as damaged, as opposed to slightly more than 50% using the old system (Blank and Roberts, 1989; Blank et al., 1988).

Widespread forest damage has since been reported in most Central European forests (United Nations Environment Program, 1987). Although there are many differences in survey methods, some crown thinning can be found throughout European forests. Initial projections of increasing damage leading to destabilization of forest ecosystems have not been supported. Severely defoliated and standing dead trees are found in stands with less affected or unaffected trees, but there has been no large-scale deforestation as observed in Eastern Europe in areas with high sulfur dioxide (SO_2) concentrations.

The 1988 FRG inventory showed that damage has been decreasing at a significant rate since 1986. This trend was most evident for Norway spruce, with damage decreasing from 24% in 1985 to about 15% in 1988. There are regional differences in this trend: damage appears to have increased in northern Germany and decreased in southern Germany. The survey report pointed out that damage can be reversed, even for markedly damaged trees, and it also cited climatic conditions as a major contributor to damage. The increase in northern Germany was thought to be largely influenced by dry and warm growing-season conditions, and the reduction in southern Germany was related to favorable conditions (Blank and Roberts, 1989).

In the early 1980s, several researchers reported growth reductions, deteriorating crown conditions, and increased mortality in high-elevation forests in northeastern North America. Siccama et al. (1982) reported that the basal area and density of red spruce in 1979 was reduced by half compared to 1964. They suggested that the decrease was not logically related to forest successional development, and they documented no specific causes.

Johnson and Siccama (1983) reported that since the early 1960s red spruce forests at high elevations in New York, Vermont, and New Hampshire have shown dieback and large reductions in basal area and density characteristic of a stress-related disease. Because of the location and topography of these forests, it was suggested that climatic stressors, such as drought, and pollutants in the form of acidic deposition and airborne trace metals may have contributed to the decline. Therefore, the decline may be due to multiple stressors, with, for example, acidic deposition enhancing drought stress or vice versa.

Hamburg and Cogbill (1988) used historical pollen counts and other historical data sources to propose that red spruce radial growth reduction, mortality, and a decrease in the size of the red spruce population since 1800 is related to a warming trend in both mean annual and mean summer temperatures. They further speculated that a natural population decline may have been aggravated by pollution or pathogens.

Millers et al. (1989) reviewed eastern hardwood forest condition over the last century. The species studied included American beech, black cherry, eastern white pine, northern red oak, shagbark hickory, sugar maple, white ash, white oak, yellow birch and yellow-poplar. Declines were reviewed to determine the location, time period, duration, areal extent, severity of damage, stated causes of decline, and possible contributing factors. Millers et al. reported that many species showed declines and mortality, and that the incidence has increased in the last few decades. Most mortality could be attributed to weather, silviculture, or insects and pathogens, and the increase in decline was attributed to more consistent reporting and to forest maturation. There was evidence of damage from point-source pollutants, but no consistent relationship between mortality and regional pollution.

Scott et al. (1984) reported large reductions in red spruce basal area at high elevations on Whiteface Mountain, NY. At lower elevations, the decline was present but at a reduced rate. Hornbeck and Smith (1985) used tree cores from a broad-scale survey of New England low-elevation dominant and codominant red spruce. They found a large reduction in tree basal area increment occurring since the early 1960s. Reports of red spruce decline are not restricted to the Northeast. Bruck (1984) and Adams et al. (1985) reported red spruce declines occurring since the 1960s in the southern Appalachian Mountains. However, red spruce growth reduction in the Great Smoky Mountains was reduced or absent at lower elevations.

The latest periodic forest survey in the Southeast, conducted by the Forest Inventory and Analysis (FIA) Research Work Unit of the USFS, reported an approximate 25% to 40% slowing of radial growth of pines in natural stands in the Piedmont and mountain regions of South Carolina and the Piedmont region of Georgia (Tansey, 1984; Sheffield and Knight, 1983; Sheffield et al., 1985). Sheffield et al. (1985) also reported diameter growth reductions for small diameter loblolly pine for Georgia and South Carolina Coastal Plain areas.

Because of these recent reports on dieback and reduced growth in eastern forests, research was expanded to include northern and eastern hardwood species from Minnesota to Pennsylvania and as far south as Tennessee. Research by McClenahen (1978) documenting changes in forest communities due to air pollutants in the Ohio River Valley indicated that species diversity generally reduced as proximity to pollution sources increased.

The potential for air pollution damage in the western United States has been well documented by Miller (1983). Miller's research during the 1970s in the San Bernardino Mountains of California indicated that high levels of ozone (O_3) were present due to easterly air flow from Los Angeles. Ponderosa pine has shown foliar injury, premature leaf fall, and decreased diameter growth. Ozone has predisposed ponderosa pine to bark beetle and root pathogen attack, and the species composition of forests in more polluted areas appears to have changed.

2.3 Natural Factors Influencing Forest Condition

Both anthropogenic factors, such as harvesting and site preparation, and natural factors, such as stand dynamics, climate, and insects and pathogens, may be responsible for changes in forests. Any effect of air pollution must be extracted from natural patterns of forest change; thus, it is necessary to understand the influence of natural factors on growth, mortality, and overall condition of forests. In this section, we discuss natural sources of variability studied in the FRP.

2.3.1 Stand Dynamics

Forest productivity is determined by the growth, mortality, and reproduction of individual trees within a forest stand. Because forest decline is often inferred from reductions in individual tree or stand growth rates, it is important to understand how stand dynamics contribute to tree and stand growth rates. The factors most closely related to productivity and often referred to as the main components of stand dynamics are (Avery and Burkhart, 1983):

1) the point in time of stand development (age);
2) the degree to which the site is occupied (stand density); and
3) site quality.

The first component, the point in time of stand development, is commonly indicated by individual tree and/or stand age. Tree and stand growth rates pass through several stages (see Figure 1). In the seedling to sapling stage, both diameter and height growth rates of trees are high. The mortality rate of seedlings is high as well. Trees move into the pole stage after a number of years (which varies by species and site), and growth may slow. As trees mature, growth rates peak and begin to decline. Later, there is virtually no height growth, and diameter growth declines. Thus, all trees and stands pass through stages of increasing and decreasing growth rates as part of normal development (Hyink and Zedaker, 1987). The age at which the various stages are reached is a function of species and stand dynamics. Loblolly pine, for instance, reaches maximum current

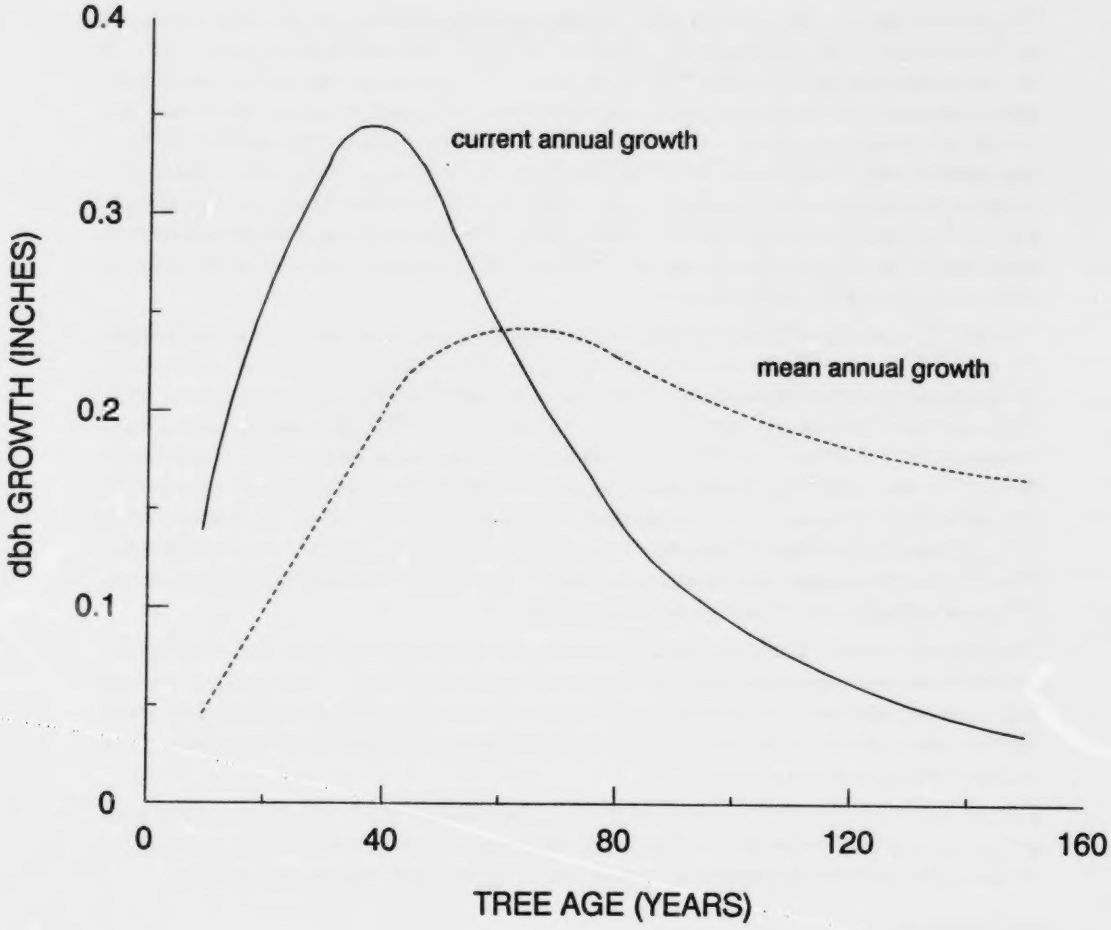


Figure 1. Comparison of current annual and mean annual growth of a tree. Current annual growth is the increase in diameter for one year, and mean annual growth is the total tree diameter divided by total age at any point in time (redrawn from Avery and Burkhart, 1983).

annual growth rate at approximately age 10. For more long-lived species, this peak growth rate can occur at age 50 or later. To understand and interpret forest data, the point in time of tree and stand development must be known.

The second component of stand dynamics, the degree to which a site is occupied, is referred to as stand density. Quantitative measures of stand density are the number of trees and the basal area or volume per unit area. Stand density reflects competition among trees. Basal area or volume per unit area of a stand increases to an asymptote that represents the biological carrying capacity for the species on the site. As this progression occurs, stand growth decreases and mortality increases.

The third component, forest site quality, is the sum of all factors affecting the site's capacity to produce tree and other vegetation. Forest productivity is generally expressed as gross volume of wood/unit area/year over the normal life of the stand. Because competition and the culmination and decrease of growth rate occur earlier on the best sites, site quality estimates are useful when evaluating whether growth rates are as expected. Site quality can be estimated directly from long-term records of stand growth and development. In many parts of the United States, tree height at various ages (referred to as the site index) is positively correlated with site productivity and is often substituted as an estimate of site quality. Site quality is also sometimes estimated indirectly by investigating the vegetation (overstory trees, indicator species) or the physical environment (soil, topography, climate).

The three components of stand dynamics appear to apply to both even- and uneven-aged forests. Structural characteristics of even-aged stands have been documented. However, comparable studies in old-growth uneven-aged stands are limited because relatively few truly uneven-aged or old-growth forests exist and because criteria for evaluating differences among uneven-aged stands are not well defined. The view that an uneven-aged stand is a collection of small even-aged patches in various phases of development was presented by Pinchot (1905) and more recently by Lorimer (1989). Because forest tree and stand productivity depend on structural characteristics, concepts based on traditional even-aged stand dynamics appear to apply to the even-aged patches that collectively constitute multi-aged stands when tree growth is summarized by age since release, not by chronological year (Lorimer and Krug, 1983).

The presence, relative abundance, and relative size of various species in the forest reflect the nature of the forest ecosystem and thus may be site quality indicators. Tree species are useful indicators because trees are long-lived, and many species are relatively unaffected by stand density. Some species have such a narrow ecological range that their occurrence indicates a particular site type or productivity.

Soil properties are also used to assess site quality. The effective depth and characteristics such as texture and pore size that affect soil moisture and aeration are important. Finally, topographic position often is related to physical soil properties that govern soil moisture and aeration.

2.3.2 Weather

The current state of the atmosphere surrounding a region defines weather. Weather is the net effect of temperature, barometric pressure, wind speed and direction, humidity, clouds, and precipitation. Because trees are long-lived perennials, they must adapt not only to long-term variations in weather, but also to wide daily and seasonal fluctuations. Extreme weather events, such as hurricanes, tornadoes, and ice storms, can produce immediate and long-term reductions in the growth and health of trees and forests through wind-throw, damage to trees and crowns, or impacts on flowering. Flowering can be disrupted by an extreme weather event; alternatively, energy put into flowering as a result of weather-induced stress can result in reduced tree growth. Sudden temperature fluctuations at critical periods of tree development can also cause physiological damage to trees, resulting in reduced health and vigor of surviving trees, as well as mortality.

2.3.3 Climate

In practical applications, climate is most often defined as the average or characteristic temperature, precipitation, and prevailing winds of a region. Implicit in this definition is the integration of weather events over time. Such a limited definition of climate is necessary because data on solar radiation, cloudiness, wind, and evaporation are available from relatively few weather stations.

Climate data from standard weather stations are used to define the macroclimate of a given region, and many valid generalizations can be drawn from the coincidence of broad climatic boundaries and the presence or behavior of forest trees (Spurr and Barnes, 1973). Annual means and sums of temperature and precipitation mask the important features of climate that influence trees. Thus, forest researchers usually summarize climate on a monthly basis. The heat and water available to plants and the daily and seasonal fluctuations of these factors are generally agreed to be the most critical climatic determinants of plant growth and species distribution (Spurr and Barnes, 1973). Climate may also act indirectly by predisposing trees to insect and disease attacks or by intensifying an already present epidemic.

Climate is often ignored in growth and yield modeling because researchers have generally assumed that climate effects average out over the long time periods that are modeled. This assumption is made because spatial and temporal weather records and analytical methods are insufficient to incorporate the information into the analysis. However, growth estimates of individual trees or stands fluctuate over time, and these fluctuations can often be attributed to changes in climate. The response to climate varies across species; in addition, within a species, the response can vary over the species' natural range, and it can differ as associated tree species vary. Holmsgaard (1955) conducted perhaps the most thorough and accurate study of annual tree rings in Europe, collecting over 78,000 measurements of ring widths on increment cores, mainly from beech and spruce in Denmark. He found that the amount of rainfall from May to July was the most important predictor of beech and spruce ring width, while summer temperature was more important for oak and silver fir. For Norway spruce, precipitation quantity was the most important of the climatic conditions of the previous year.

2.3.4 Insects

Forest insects usually exist at low levels throughout forests of the world. Most insects are generally believed to be a secondary cause of growth rate decline or mortality. Some insect epidemics result when trees are weakened by extreme weather or climate events; for example, lightning-damaged conifers may attract bark beetles, and their attacks on single trees can then lead to mass attacks on surrounding healthy trees (Spurr and Barnes, 1973).

The view that insect epidemics are secondary in nature and are triggered by other primary causes is not universal, however. Gordon (1984) suggests that spruce-fir forests and spruce budworms evolved together, and that these insects are an essential factor in maintaining spruce in these stands. Extensive outbreaks of some insects may be part of the ecology of spruce-fir and pine forests, thus forming an important part of growth and renewal cycles in eastern forests (Baskerville, 1975).

The introduction of nonindigenous insects to a region can result in reduced vigor and extensive mortality. The balsam woolly adelgid, an insect introduced from Europe, has increased mortality rates of Fraser fir throughout the southern Appalachian Mountains. The gypsy moth is another example of an introduced insect causing reduced growth and increased mortality, especially for oak species in eastern states.

2.3.5 Pathogens

Forest tree disease is an unfavorable departure from normal plant function or form, caused by a pathogenic agent or unfavorable environment. Pathogens can affect almost any part of the tree: needles, roots, main stem, branches, and cambium. Historical examples of diseases causing mass decline of a forest tree species are well documented. For example, white pine blister rust, caused by the fungus Cronartium ribicola, was introduced into North America from Europe about 1900, causing widespread growth loss and mortality. Where conditions for blister rust development are favorable, it has virtually eliminated susceptible pine species.

Some past declines of forest trees have been incorrectly attributed to other causes when a pathogen was the main culprit. Such was the case in Europe where root investigations by Wagenknecht (1960) showed that the decisive factor in "arable land sickness," which decimates afforested stands of Scots pine on agricultural land, was not the impediment of root development but rather was *Fomes annosus* (now called *Heterobasidion annosum*), which attacks the roots. A large reduction in height and diameter growth resulted, with eventual death for some trees.

Tree diseases have been associated with air pollutants. Ozone is known to cause a needle disease of ponderosa pine that results in decreased growth, needle drop, and, in some cases, death. Susceptibility to insects and disease can vary both within and among species. For example, some eastern white pine genotypes are highly sensitive to air pollutants (Karnosky, 1980). Eastern white pine also suffers from chlorotic dwarf (Dochinger and Seliskar, 1970) and post-emergence tipburn (Berry and Hepting, 1964).

3 METHODS AND MATERIALS

Research conducted to investigate the two scientific questions addressed in this document used the epidemiological approach, characterized by broad surveys of forest condition and a search for patterns that may be related to natural factors or atmospheric deposition. Several types of forest survey information are discussed in this section. Methodologies for determining atmospheric deposition and ozone exposure also are discussed.

3.1 Dendrochronological Analysis

The techniques and limitations of dendrochronological (i.e., dated tree ring) analysis are discussed in detail in Appendix A. Thus, only a brief summary of the technique is given here.

First, tree core samples are taken from the population of interest. The obtained tree rings are then measured and crossdated; that is, annual ring widths for each tree are matched with calendar year by comparing each sample to a known or master ring width series (Fritts, 1976). Next, ring widths are modeled to determine how environmental factors, both natural and anthropogenic, affected tree ring growth. Both endogenous disturbances (those affecting a single tree or a subset of trees in a stand) and exogenous disturbances (those affecting radial growth across stands) must be considered. To assess the effects of pollutants, the pollution signal must be separated from all other factors. Linear aggregate and multiplicative models have been used.

Data are usually standardized to estimate and remove the natural growth trend from the raw ring-width sequences for each tree. Next, a site chronology is formed by averaging the indexed sequences of all trees for a site. If there is autocorrelation, (i.e., if year-to-year observations are correlated), this systematic component may be removed by autoregressive moving average (ARMA) modeling. This step separates autoregressive disturbances from sampling error.

The residual sequence should now consist of the common signals of climate, exogenous disturbances, and random error. The estimated change in growth due to climate is then modeled, usually by using multiple linear regression to predict growth from temperature and precipitation. It may then be possible, after allowing for the effects of age and stand disturbances, to predict ring widths of trees thought to be affected by pollutants. Consistent overestimation (or underestimation) could be evidence of a negative (or positive) effect of pollution and provide a measure of its magnitude.

3.2 Transect Surveys

Transects of varying lengths were used to sample tree vigor class at a number of mountain sites. Transects are generally run at different elevations so that elevational differences in forest conditions can be quantified. In sampling, the tree nearest to the transect line that exceeds a lower size class limit is sampled. In studies cited in this document, crown classification and live/dead status were the measures recorded for each sample tree/transect. Transects or sample trees are not identically remeasured over time, as initial transects are not usually permanently marked. Although the same forest stands are represented, this difference between surveys is a potentially large source of variation.

3.3 Permanent Study Remeasurement Data

When sampling trees to determine forest condition over time, plots of a fixed area are often used. Both the number of plots required and the size of each plot depend on the size and variability of the population, required sampling precision, and cost (Husch et al., 1972). In general, the greater the desired precision and the more varied the forest composition, the more sampling that is required. Small plot areas may increase precision since more plots can be established and thus the number of independent samples increases. However, in heterogeneous forests, small areas may not be representative and may increase variation in the data. Avery and Burkhart (1983) suggest that optimum plot size for standing inventories of second-growth forests is from 0.02 to 0.10 ha; plots of 0.04 to 0.10 ha are common.

The data obtained from plots can include both plot and individual tree measures. Examples of plot data are location, cover type, stand size and condition, stand age, stocking, slope, soil classification, and understory vegetation. Individual tree measures include species, diameter at breast height (dbh), height, crown class, vigor, diameter growth, and mortality.

To evaluate changes in forest condition over time, permanent plots are often established. With this method, the same trees at each plot are remeasured at successive points in time. The sampling error associated with remeasurement and growth and change estimates is likely to be lower than the error associated with independent samples (Avery and Burkhart, 1983). Permanent plots are chosen using two criteria: 1) plots must be representative of the forest population for which inferences will be made; and 2) plots must be subject to the same management practices as the unsampled portion of the forest. The sampling procedures and measurements must be consistent at each measurement interval to reliably determine changes in forest condition over time. Remeasurements are typically made at 2- to 10-year intervals. Long intermeasurement periods can mask important changes in the rate of growth and thus should be avoided.

3.4 Forest Inventory and Analysis Data

The Forest Inventory and Analysis (FIA) Research Work Unit of the Southeastern Forest Experiment Station inventories the forests of Florida, Georgia, North Carolina, South Carolina, and Virginia (Sheffield et al., 1985). FIA data from Pennsylvania and Vermont (Northeastern FIA) and from Minnesota, Wisconsin, and Michigan (Northcentral FIA) are also collected and are presented in this document. The FIA inventories across the United States are conducted approximately every 10 years to measure changes in the population. However, measurement intervals often range from 10 to 20 years. In the Southeast, the first inventory was initiated in 1933, and the fifth was completed in 1985. There are approximately 25,000 permanent FIA plots in this region, each representing an average of 1,375 ha.

For FIA remeasurement data used in the FRP (which includes data from Southeast, Northeast, and Northcentral FIA), Bitterlich horizontal point sampling (Avery and Burkhart, 1983) is used. This method is a form of selection with probability proportional to size. From a given sampling point, trees that are close enough and have a dbh large enough to exceed a fixed angle of view projected by an angle gauge are selected as sample trees. The gauge angle is determined by the basal area factor (BAF), which is the ft^2/acre that a tree represents. The BAF is usually determined according to the size of trees in the region to be sampled. This system is widely used in forestry, and it provides for rapid and unbiased estimates of stand conditions. It is analogous to using concentric variable radius plots where the plot radius is a function of tree diameter. This system is also known as variable radius plot sampling or prism plots. In subsequent surveys, the same trees are remeasured if they are alive, and other trees now large enough to be included are measured and added to the inventory.

In the Southeast, inventory methods and sampling techniques have changed over the years. In the first two inventories, trees were bored to determine diameter growth rates. The first inventory followed compass lines 16 km apart, with 0.10-ha fixed-area sample plots located every 200 m. In the second survey, plots were randomly selected from a systematic grid placed on aerial photographs. Permanent plots were established so they could be remeasured. In the third survey, a transition was made from the increment core technique to diameter remeasurements, and Bitterlich horizontal point sampling was initiated (Avery and Burkhart, 1983).

Stand basal area growth is calculated from the diameter growth of trees in each inventory. Stand basal area growth is calculated from the diameter growth of trees that occurs from one inventory to the next. Net basal area growth is usually of greatest interest and is defined as: survivor growth + ingrowth - mortality - cut; where survivor growth is the growth of live trees measured at both inventories; ingrowth is the basal area of trees growing into a measurable size; mortality is the basal area of trees that died between the two inventories; and cut is the basal area of trees harvested between the two inventories (Beers, 1962).

To analyze individual tree growth from the Southeast FIA data, average annual radial increment (AARI) is calculated for each tree and then a mean AARI is calculated for each diameter class and species. Even though the same trees are remeasured, some trees die or are cut, new trees are included, and most trees grow into larger diameter classes. Comparison of mean AARIs for successive time periods thus indicates how trees of the same diameter class grew in each period, not how growth of the same trees might have changed.

The nonindustrial private lands of the Southeast surveyed by FIA are not usually as intensively managed as are industrial private lands, and trees of all crown classes are included. High percentages of intermediate and overtopped trees in the smaller diameter classes suppress growth values. Thus, growth curves are not typical of those derived for intensively managed stands. The changes in FIA inventory design over the years make comparisons of individual-tree growth rates difficult. Individual tree growth data from the earlier surveys are limited; thus, reports of growth reductions are based on comparisons of average growth rates from the fourth and fifth survey periods only. Additional difficulties arise in determining possible causes of tree decline because some important tree, stand, and site information is not available for all plots. Omitted information can include crown position, crown ratio, competing basal area of trees and shrubs, site quality and prior land use. However, some of these measures can be added to the sampling procedure.

3.5 Spatial Study Data

Spatial studies are conducted along known deposition gradients spanning large geographic areas. Fixed area plots are sampled at sites along the gradient to identify patterns in forest response to natural factors and pollutants; in particular, the responses at either end of the gradient are compared to determine the magnitude of change. To minimize variation in response due to differences in stand characteristics, analogous plots are selected to have similar stand characteristics. Responses measured along the gradient include tree growth, tree mortality, crown vigor, and precipitation, throughfall, soil, and litterfall chemistry. Spatial studies have been conducted along gradients in the Great Lakes region (Minnesota to Michigan), in the Ohio Valley, (Arkansas to Ohio), and in Pennsylvania.

3.6 Aerial Photography and GIS

Maps, in conjunction with software designed to analyze large data bases of geographic information, have been used to evaluate forest condition for large geographic areas in the Northeast (Miller-Weeks, 1989) and in the southern Appalachians to map standing dead. For example, Dull et al. (1988) estimated standing dead in Southern Appalachian spruce-fir forests using aerial photography, ground plots, maps, and Geographic Information System (GIS) software; this study will be described to illustrate the technique, since most studies of this type are similar.

First, existing stand type maps for the area were used to identify known spruce-fir stands. Flight lines were then plotted to include all spruce-fir stands. During the summer of 1984, aerial color infrared photographs were taken at a scale and overlap that allowed individual trees to be studied stereoscopically under magnification. The entire spruce-fir population area was photographed, and each photograph covered approximately 750 ha.

Next, photo interpreters classified the area within each of six previously defined geographic regions by standing dead and they determined the spruce-fir stand boundaries. Standing dead classes were then randomly sampled by plots in proportion to the area of spruce-fir affected. For each plot selected in this sampling, the total number of dead spruce and fir trees and the total number of spruce and fir trees were recorded. A random ground subsample of the selected aerial photo plots was also performed, providing data on species, dbh, crown condition, total height, and crown position for each sampled tree, as well as site information. Spruce and fir were not separated when using the aerial photographs, but these species were distinguished in the ground samples.

Standard class boundaries were then traced with a digitizing tablet, and the data were entered into a digital data base of coordinates. Next, these map files were placed in a geographic analysis system and projected onto maps that could be viewed on a computer screen or printed out. Users retrieved, analyzed, and displayed these maps, and maps could be overlaid on each other. Analyses conducted by Dull et al. included location and area of spruce-fir, distribution of standing dead, comparison of present and past standing dead, and associations of other variables with standing dead (e.g., elevation, balsam woolly adelgid, and disturbance history).

3.7 Wet Deposition Data

While some researchers have used survey data to evaluate forest growth and condition, other research has been conducted to assess trends across the United States in atmospheric deposition of pollutants and exposure to ozone. These data are presented in this report, and the methods used to obtain the data are described here.

Deposition of hydrogen (H^+), sulfate (SO_4^{2-}), ammonium (NH_4^+), and nitrate (NO_3^-) ions from the atmosphere occurs through wet and dry processes. Wet deposition occurs via both precipitation and cloud water interception by forest canopies. In this report, annual summaries of precipitation chemistry from the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) are used to describe spatial (horizontal) variation in wet deposition (NADP, 1988; Olsen and Slavich, 1986). Cloud water chemistry data from the Mountain Cloud Chemistry Project (MCCP) also are presented to describe elevational (vertical) variation (Mohnen, 1988a,b; Vong, 1989a).

Elevational variation in deposition is primarily related to the amount of time a forest is immersed in clouds. In the East, at elevations below about 1000 m, wet deposition is primarily due to precipitation, while above 1000 m cloud water chemical deposition may equal or exceed precipitation deposition. Cloud frequencies and chemical flux contributions from cloud water have not yet been quantified for the West.

Either wet deposition or concentrations of H^+, SO_4^{2-}, NO_3^-, and NH_4^+ may be relevant to describe acidic precipitation, depending on the ecological effect of interest. For example, foliar leaching in spruce needles might be related to chemical concentrations in acidic precipitation, while soil buffering processes might reflect wet deposition. Because observed precipitation concentrations of H^+ (NADP, 1988) are consistently lower than those found to harm seedlings (C. Peterson et al., 1989), precipitation deposition, not concentration, is considered relevant to forest effects. However, because of the much higher chemical concentrations (lower pH) in cloud water, both cloud water chemical concentrations and deposition are considered relevant to forest effects.

Regional estimates such as isopleth maps typically are used to show the magnitude and extent of acidic deposition and to locate areas of high or low deposition. Spatial interpolation is necessary to generate contour maps of wet deposition and to estimate data for non-monitored locations. Recognizing that there are numerous equally valid methods for presenting these data, a regional map with bars was used because it best demonstrates annual deposition, its variability, and gradients over regional scales (< 100 km). Contour maps require interpolation assumptions and can mask some small-scale variability, but they are also very useful for displaying large-scale spatial trends (> 100 km)(Vong et al., 1989; Olsen, 1989).

Wet deposition data are discussed for six forested regions in this report: 1) the Northeast; 2) the Southeast; 3) the upper Midwest; 4) the lower Midwest; 5) the Northwest; and 6) the Southwest. For these regions, rather than interpolate data, measurements from NADP/NTN sites were used to illustrate spatial patterns. A site was selected if:

1) it was in a forested region studied by the FRP; and

2) at least two annual wet deposition summaries were available for 1985-1987.

To produce summaries, multiannual averages from only NADP/NTN sites were specified to ensure data comparability and reduce temporal variability. Contour maps for individual years are available elsewhere (summarized by Vong et al., 1989).

Because cloud water deposition occurs primarily at high elevations, the spatial patterns in total wet deposition (i.e., cloud plus precipitation) are more complicated than would be inferred from large-scale maps of precipitation deposition. Therefore, two patterns are discussed where data are available:

1) spatial variation in precipitation deposition (by longitude and latitude);

2) the distribution of cloudiness and resulting spatial variation in cloud water chemical deposition (vertical and horizontal patterns).

Maps of precipitation deposition are presented in the regional deposition sections in eq/ha/yr for four ions: H^+, SO_4^{2-}, NO_3^-, and NH_4^+. Table 1 shows how to convert these units to kg/ha/yr.

Table 1. Conversion between units of annual wet deposition.

100 eq/ha/yr H^+ = 0.1 kg/ha/yr
100 eq/ha/yr SO_4^{2-} = 4.8 kg/ha/yr
100 eq/ha/yr NO_3^- = 6.2 kg/ha/yr
100 eq/ha/yr NH_4^+ = 1.8 kg/ha/yr

3.8 Ozone Data

As for atmospheric deposition, research has been conducted to describe trends in ozone exposure in regions across the United States. Tropospheric ozone (O_3) is a secondary pollutant formed when nitrogen oxides (NO_x) react with non-methane hydrocarbons and other organics in the presence of sunlight (e.g., Finlayson-Pitts and Pitts, 1986). The sources for O_3 are thus related to those for NO_x and hydrocarbons: Anthropogenic emissions in the United States primarily result from transportation, fossil fuel combustion, petroleum refining, oil and gas production, and organic solvents (Finlayson-Pitts and Pitts, 1986). The air pollution potential of a region is determined by emission and chemical transformation rates, and by weather conditions such as temperature inversions. Pollutants can be trapped beneath an inversion layer, prohibiting their dispersal, or they can become trapped above an inversion layer, allowing transport over large distances prior to deposition.

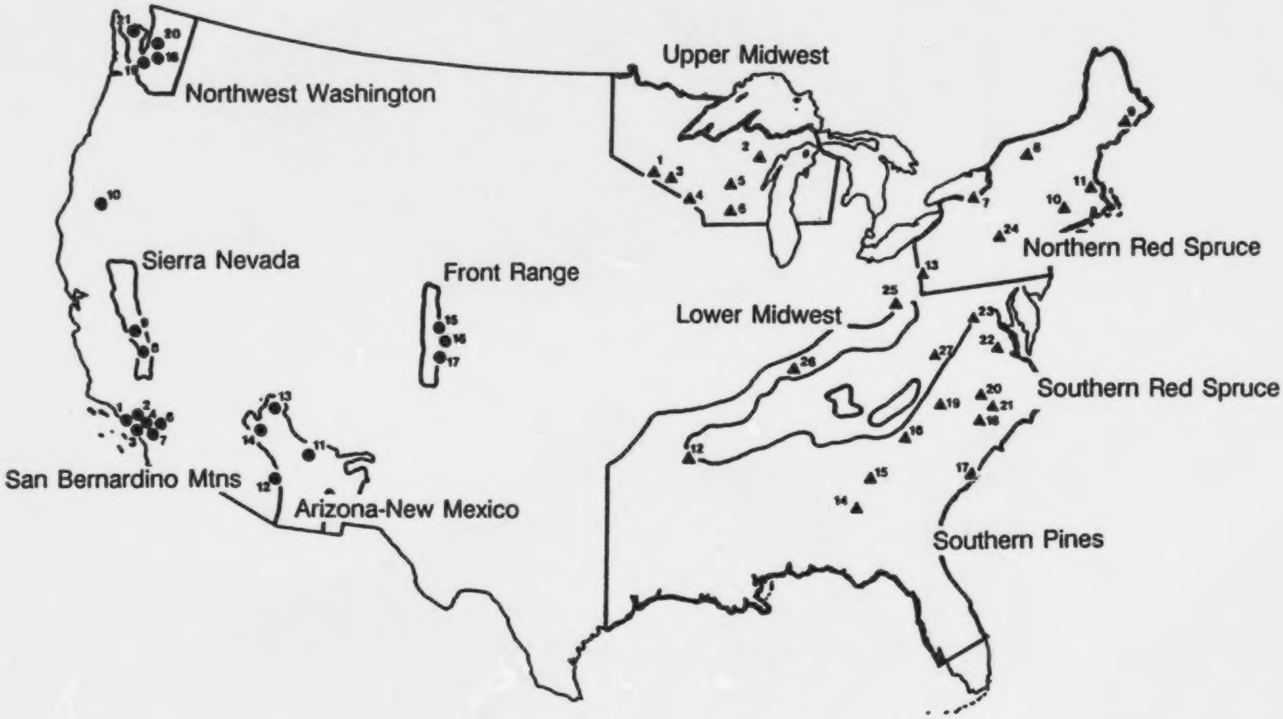
Ozone is the only regionally dispersed pollutant known to injure foliage and to predispose trees to mortality (Woodman, 1987). Ozone injury to the coniferous forest species is well documented (e.g., Miller et al., 1963; Miller, 1973, 1983; Peterson et al., 1987; Peterson and Arbaugh, 1987; Temple, 1988). Regional and national trends in O_3 exposure are reported in this document. However, because O_3 levels are not known for the forested sites in FRP studies, formal correlations of O_3 levels and forest growth and conditions are not possible in this report.

Several indices of O_3 exposure are typically used, such as simple 24-hour means and cumulative or weighted sums. When relating growth trends to pollutant exposure across large spatial areas, these indices may not reflect subtle differences in O_3 exposure (and consequently response) at different places. For example, an analysis of O_3 monitoring data along a gradient from Arkansas to Ohio revealed no significant differences in growing-season 7-hour mean concentrations for 1978-1985 (Foster, 1989). However, diurnal patterns for three sites along this gradient during the summer differed substantially: In 1988, peak daily concentrations were much higher and minimum daily concentrations much lower at the eastern end of the gradient than at the western end. Because the "physiologically effective" dose for O_3 is the integrated flux into leaves via the stomata, the correspondence between diurnal patterns of stomatal conductance and O_3 concentration determines the effective dose (Guderian, 1985; Krupa and Teng, 1982; Tingey and Taylor, 1982). Thus, because differences in diurnal O_3 patterns may produce different physiological responses, an index that incorporates both the temporal aspect and the magnitude of O_3 concentration is desirable.

Because of such considerations, O_3 data from the EPA Aerometric Information Retrieval System (AIRS, previously SAROAD) database discussed in the regional sections of this report were analyzed by Böhm and Vandetta (1989) to identify diurnal and seasonal patterns. Figure 2 shows the forested regions studied. There were generally too few data to investigate temporal trends. Average seasonal and diurnal patterns in hourly O_3 concentrations were constructed by combining all data for a given hour and month across years. If 75% or more of total available hours were represented, the hourly median was calculated (O_3^{med}). The worst-case scenario, defined as the second highest concentration ever recorded for a specific month, was also determined (O_3^{worst}).

Since the formation of O_3 is related to meteorology, which can vary considerably from year to year, the median can be influenced by the sampling period. To reduce the likelihood of such influence, periods longer than three years were usually used for averaging. Worst-case scenarios can also be influenced by sampling period, but at worst they will underestimate extreme conditions. Seasonal and diurnal patterns in O_3^{med} and O_3^{worst} do not reflect actual patterns, but are rather 50th percentile and approximately 99th percentile surfaces.

Graphs can provide information on subtle differences in exposures across hours of days within months and across months within hours of days that may not be shown by averages or in cumulative exposure data; in particular, they may show the occurrence of extreme conditions as well as differences across sites. By preserving some degree of temporal flexibility, O_3 concentrations for those periods when plants may be most susceptible can be identified. In regional discussions, figures are presented where they are important to illustrate a result; otherwise, data are shown in tables. In the figures, data are presented in three dimensions: hour of day, month, and ozone concentration (see Figure 3). To aid interpretation, isolines connect respective points with 60, 80, 100, and 120 ppb.


Northwest Washington
21
20
18
11
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Sierra Nevada
9
8
Front Range
15
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17
1
2
3
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6
7
San Bernardino Mtns
13
14
11
12
Arizona-New Mexico
Upper Midwest
1
2
3
4
5
6
9
8
7
11
10
24
Northern Red Spruce
13
25
Lower Midwest
23
22
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26
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19
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Southern Red Spruce
Southern Pines

Figure 2. Location of the forest type study areas, and the ozone and wet deposition study sites. Numbers refer to site numbers in Tables 3 and 9.

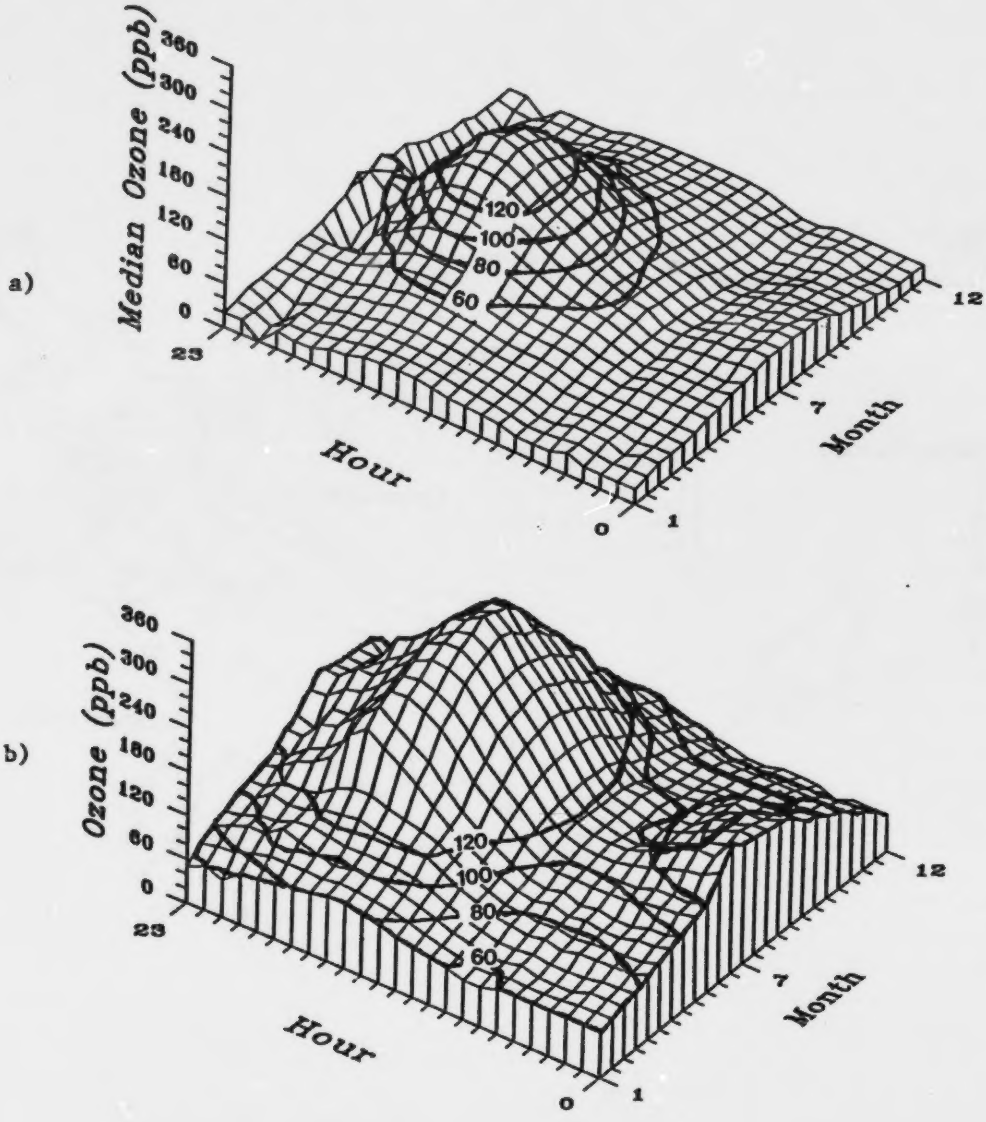

Figure 3. Examples of the three-dimensional plots of median and worst-case O_3 concentrations (ppb) used to analyze data presented in this report. Data are from San Bernardino County, CA. a) hourly median O_3 concentration; b) composite of second maximum hourly values.

4 RESULTS

Results for individual studies are presented by forest type. In each section, forest condition, wet deposition, and ozone exposure data are discussed. Where possible, information on soils, insects, and pathogens is also presented.

4.1 Northern Red Spruce

Spruce-fir forest occurs on approximately 4.5 million ha in New England and New York (Gibbs, 1979). Of this, about 3.2 million ha are in Maine. In the Adirondacks and northern Appalachians, the areal extent of high-elevation spruce-fir forests is estimated at 132,000 ha (Eagar, 1989, cited in Barnard et al., 1989). A number of FRP studies have been conducted at selected mountains in the Northeast that support spruce populations. For low elevations in New England, red spruce are represented by FIA data.

4.1.1 Forest Condition

Standing Dead and Mortality

Observational studies indicate increases in mortality of red spruce at high elevations since the early 1960s (Siccama et al., 1982; Scott et al., 1984; Johnson et al., 1988). A review of the literature indicates at least one other major period of red spruce mortality in the Northeast in the last 100 years: A large episode of red spruce mortality occurred across northern New England and New York from about 1871 to 1885 (Hopkins, 1901), during which it was estimated that 50% of the mature spruce in the Adirondacks died (Seventh Report of the New York State Forest Commission, 1891). The exact causes are not agreed upon, but the spruce beetle is known to have contributed (Hopkins, 1901).

Johnson et al. (1988) reported that between the mid-1960s and mid-1980s, red spruce died at high rates in the Adirondack, Green, and White Mountains. At 1200 to 1300 m, 30% to 60% of the red spruce basal area was standing dead, but a smaller proportion was standing dead below 800 m.

Battles et al. (1989) reported a trend towards a lower percentage of live stems at higher elevations for all species on Whiteface Mountain, NY. For red spruce, the percent of live stems below 1000 m was 72%. Above 1000 m, the percent of live stems decreased to 43%, and the percentage of live stems decreased as elevation increased from 1000 to 1200 m and above. The percentage of dead red spruce was greater in wind-stressed areas than in wind-protected areas. However, on wind-protected aspects, the percent of dead spruce exceeded that of other species by approximately 100%. Thus, the authors concluded that wind stress alone was not the sole cause of the mortality. Comparing 1982 with 1987, Battles et al. showed that 25% of the canopy red spruce greater than 10 cm dbh died over that five-year period, and they concluded that this mortality rate was more rapid than expected for mature red spruce.

Silver et al. (1988) surveyed red spruce trees on three mountains in the Northeast (Mt. Washington, NH; Mt. Mansfield, VT; and Whiteface Mountain, NY). Two survey transects at the same elevations in 1982 and 1987 were used to compare percent standing dead red spruce at three elevations (below 900 m, 900-1049 m, and above 1049 m). Because the 1982 transects were not permanently marked, the 1987 transects did not include the identical trees surveyed in 1982. On Whiteface Mountain, there were more standing dead trees at all three elevations in 1987 than in 1982 ($p < 0.05$). Percent standing dead at Mt. Mansfield increased from 35% to 37% from 1982 to 1986 and at Mt. Washington, percent standing dead increased from 20% to 28% from 1982 to 1987. However, neither increase was statistically significant ($p > 0.05$) (see Table 2).

Table 2. Percent dead along sampling transects in 1982 and 1987 (from Silver et al., 1988).

	% dead		sample size[a]		est. ann. mort. rate[b]		
	1982	1987	1982	1987	(%/yr)	z[c]	z[c]
Whiteface Mtn							
<900 m	0.120	0.267	166	176	3.33	3.28	**
900-1049 m	0.146	0.443	104	97	6.97	4.49	**
>1049 m	0.326	0.500	92	64	5.16	2.02	*
Mt. Mansfield							
<900 m	0.295	0.339	112	115	1.26	0.58	n.s.
950-1049 m	0.373	0.327	102	98	-1.47	-0.83	n.s.
>1049 m	0.315	0.486	54	35	4.99	1.40	n.s.
Mt. Washington							
<900 m	0.229	0.278	170	169	1.26	0.91	n.s.
950-1049 m	0.202	0.272	119	114	1.76	1.11	n.s.
>1049 m	0.188	0.300	96	80	2.77	1.57	n.s.

[a] total number of trees surveyed.

[b] estimated as :

$$\frac{(\%\ \text{dead 1987}) - (\%\ \text{dead 1982}) \times 100}{5 \times \%\ \text{alive 1982}}$$

[c] * $p < .05$

** $p < .01$

Averaging data from the three mountains, the percent of dead red spruce (> 10 cm dbh) increased from 26% in 1982 to 37% in 1987.

Peart et al. (1988) determined the percent of standing dead (basal area) for red spruce and balsam fir at three elevations (840, 990, and 1200 m) on Mt. Moosilauke, NH. The percent dead red spruce on the east-facing slope averaged 34% and changed minimally with elevation. On the west slope, red spruce standing dead averaged 29%, with the highest percentage (41%) at the middle elevation. For balsam fir, 17% and 16% of basal area were standing dead for east and west aspects, respectively.

Friedland (1989) compared proportions of standing dead red spruce for 19 mountains. The mountains were randomly chosen from all those with elevations greater than 1220 m in five areas along a west-to-east transect of the New York Adirondacks and the Appalachians of Vermont, New Hampshire, and Maine (see Figure 4). Dry and wet deposition of pollutants are higher in the western than in the eastern part of the transect, but uncertainty in estimates of cloud water deposition precludes conclusions about an actual gradient in total deposition (see Section 4.1.2). At each mountain, six non-permanent plots, 30 m apart, were established on the west and east side at four elevations (950, 1000, 1050, and 1100 m), for a total of 48 plots per mountain. At each plot, two types of surveys were conducted. In a visual assessment survey, the four canopy red spruce trees (those receiving direct sunlight to the top of the crown) nearest to plot center were measured and assigned a health rating. In the second type of survey, stand basal area was estimated for each plot using a 2.5 metric BAF. Species, dbh, and live/dead status were recorded for each tree selected in both surveys.

In the visual assessment survey (Friedland, 1989), the proportion of recent dead red spruce was greater in the Adirondack and Green Mountains than in mountains to the east (30% versus 14%; $p < 0.01$). Total standing dead red spruce averaged 42% compared with 13% for all other species (see Figure 5). The Adirondack and Green Mountains showed increased standing dead red spruce with increasing elevation. Percent standing dead of canopy red spruce was as high as 60% at sites above 1100 m in the Green Mountains. In the eastern mountains, there was no increase in standing dead red spruce with elevation. From the basal area survey, if only canopy red spruce are considered, the trend of decreasing dead to the east is apparent, although the trend was not statistically significant ($p = 0.05$) because there were 50% fewer canopy trees in the sample than there were in the visual assessment survey. Friedland (1989) suggested that this difference in results from the two surveys may be due to the presence of understory and suppressed trees as well as canopy trees in the basal area survey. These results seem to suggest that dying red spruce in the canopy are being replaced by trees lower in the canopy, because the basal area survey of all trees on the plot suggested no increase in red spruce standing dead.

Johnson et al. (1988) hypothesized that climatic extremes led to current levels of standing dead and decreased annual ring widths of red spruce since the early to mid-1960s. The authors suggested that winter injury over several winters resulted in annual losses of foliage and buds and served to synchronize and initiate increased mortality and reduced radial growth (see Figure 6). Several observational studies suggest that winter injury to red spruce has caused red spruce death in years following the event. Curry and Church (1952) reported on a winter injury event that occurred in the Adirondacks during the winter of 1947-1948. They attributed the injury to desiccation of needles. Starting in April of 1948, with subsequent visits in June of 1948 and October of 1949, Curry and Church monitored 25 red spruce that sustained winter injury. From the last measurement in 1949, they found that of the 10 trees with more than 75% defoliation, six had died and four appeared to be dying slowly. None of the other 15 trees, which had initial defoliation of 25% to 75%, had died; seven were described as recovering, and eight appeared

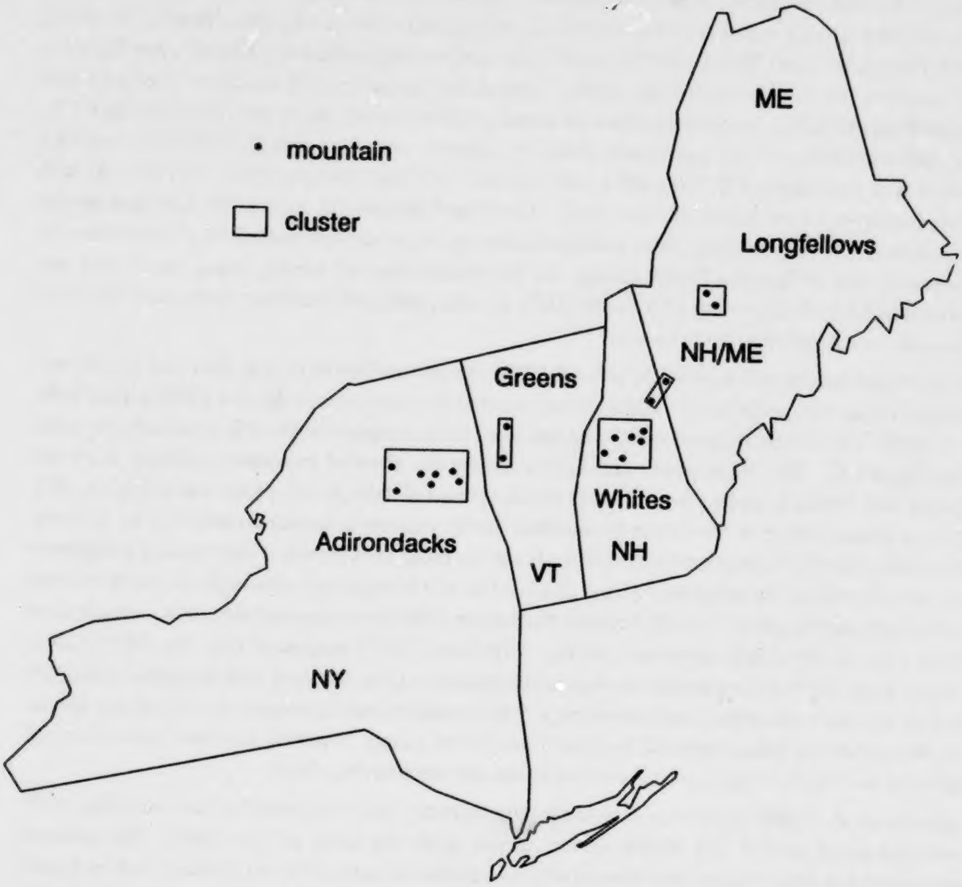
• mountain
☐ cluster
ME
Longfellows
NH/ME
Greens
Whites
Adirondacks
NH
VT
NY

Figure 4. Locations of 19 mountains and 5 clusters in New York and New England used to estimate regional variation in standing dead red spruce (redrawn from Friedland, 1989).

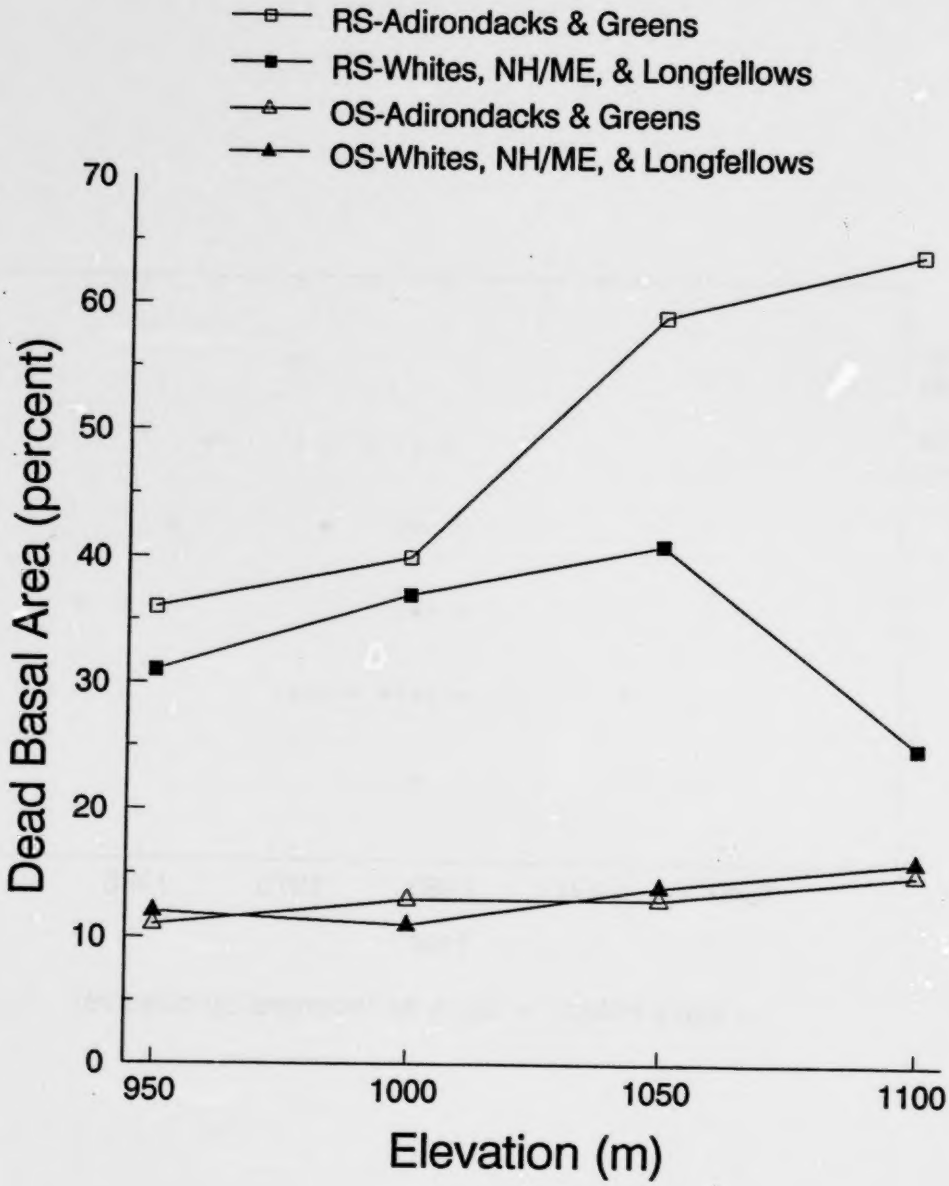

Figure 5. Comparison of percent standing dead basal area for red spruce (RS) and other speices (OS) by mountain cluster and elevation (redrawn from Friedland, 1989).

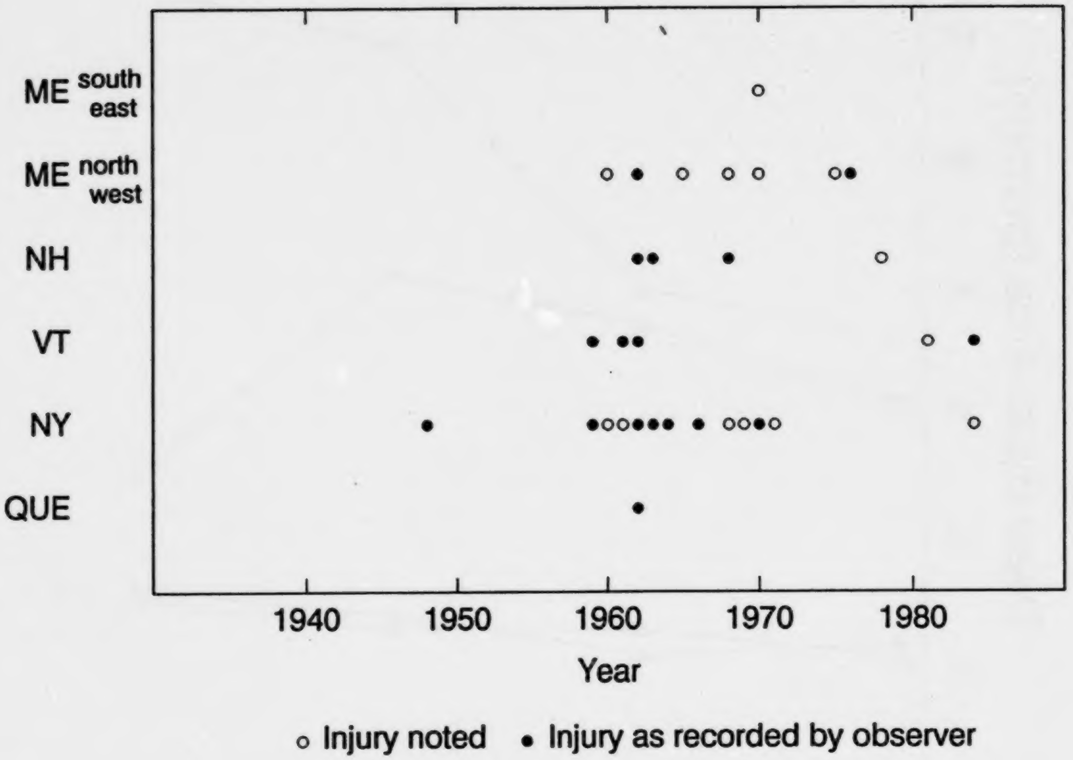

Figure 6. Reports of winter injury (drying, burn, kill, etc.) where red spruce foliage was observed in early spring. The winter injury events are generally widespread within a geographic area (redrawn from Johnson et al., 1988).

unlikely to recover. The authors noted that there was considerable mortality of saplings from the winter injury event.

Reports noting severe foliar damage to red spruce from winter injury suggest that from 1959 to 1964 winter injury occurred more frequently than had been previously documented at high elevations in the Northeast. The Vermont Department of Forest and Parks (1959) reported that large areas of red spruce were affected by severe winter drying. The same agency reported severe winter drying of red spruce at high elevations in northern Vermont during the winters of 1961 and 1962. Patches of high-elevation spruce in western Maine and New Hampshire were apparently killed by severe winter conditions from 1962 to 1964 (Tegethoff, 1964).

In the White Mountains, reports of winter injury and subsequent mortality have been documented for the early 1960s (Wheeler, 1965; Kelso 1965). In the mountains of west-central Maine, reports of winter injury in 1962 and subsequent death of red spruce along crests of many mountains was reported by Stark (cited in Johnson et al., 1988). Spruce of all ages and competitive status were affected, and the damage was largely on current year foliage. Winter drying was observed in 1968, with first- and second-year needles missing. Mountain sites or areas of exposed and thin soils were most heavily affected.

DeHayes et al. (1989) observed foliar winter injury of red spruce provenances in a plantation in Colebrook, NH. In 1986, severely injured branches were tagged for reexamination in July of that year. DeHayes et al. observed injury only on current-year needles. More than 96% of the tagged branches exhibited terminal and internodal bud break, and apparently healthy new shoots were produced despite complete loss of previous-year needles. DeHayes et al. (1989) performed laboratory freezing experiments of red spruce plant material from the provenance plantation in Colebrook, NH. They concluded that current-year needles are less cold tolerant than older needles, and that cold tolerance of current-year needles in December and January is barely sufficient to protect them from cold temperature events. The authors suggested that observations of winter injury over the last 30 years may be due to freezing rather than desiccation.

Research on winter injury thus indicates that some forms of winter injury that result in foliar damage and sometimes bud death have been associated with death of red spruce in the Adirondacks and other high elevations in New England. The injury and subsequent mortality episodes appear to have been most severe from 1962-1966. In none of these mortality episodes did the observers attribute spruce death to biotic agents. In the 1980s, most winter injury events have been confined to current-year needles. Injury to one- and two-year old needles has been observed at Camels Hump, VT, and Whiteface Mountain, NY, with 95% of the injury occurring on first-year needles (A.H. Johnson, personal communication).

Growth

In addition to studies of standing dead red spruce at high elevations, radial growth rates of red spruce trees have been studied at both high and low elevations. Johnson and Siccama (1983) reported a decrease in radial increment of red spruce in the mid-1960s for Whiteface Mountain, NY, Mt. Mansfield, VT, and Mt. Washington, NH. In the early to mid-1960s, 40% of sampled trees showed a shift to narrow ring increments that continued through 1981. Johnson and Siccama suggested that these growth patterns resulted from a period of intense stress that triggered a lasting loss of vigor, and that the growth reductions represent the initiation of decline in these trees. Drought was suggested as an initiating factor of reduced radial growth, but drought by itself was thought to be insufficient to cause a continued reduction of radial growth. Acidic deposition was suggested as an additional stress.

Hornbeck and his colleagues conducted an initial study on the status of red spruce at low elevations (below 800 m) across five northeastern regions: north and south Maine, New

Hampshire, Vermont, and New York (Hornbeck et al., 1986, 1987). The data were from 3,001 cores, one per tree, from three sources: 2,387 trees from a USFS 10-year forest inventory in Maine, New Hampshire, and Vermont; 445 trees from the Adirondack, Green, and White Mountains; and 169 trees from the White Mountain National Forest (NF). Four or fewer trees were sampled at each site. Tree age ranged from 14 to 285 years; 72% were 40 to 100 years old. More than half the sampled trees were from northern Maine, which was the area most severely infested by spruce budworm from the early 1970s through 1985.

Hornbeck and Smith (1985) averaged basal area increments over all sampled trees within each of the five regions by calendar year. They reported increasing mean basal area increments up to the early 1960s, followed by decreasing mean increments (see Figure 7). Federer and Hornbeck (1987a,b) suggested that maturation may explain this pattern, and they reported that a large percentage of stands originated and matured at approximately the same time throughout New England.

The maturation argument can be made to some degree for Maine and northern New Hampshire and Vermont because of a severe spruce budworm outbreak from 1910-1920. In this outbreak, one-half to two-thirds of the northeastern spruce-fir forest at low elevations died and subsequently regenerated (Seymour, 1985). Prior to the budworm epidemic, logging occurred over large areas of spruce and fir from 1900 to 1920, thus releasing large areas of advanced regeneration at approximately the same time as the budworm epidemic. Post-budworm stands were characterized by two broad age classes: remnants of stands that survived both the budworm and cuttings, and a new age class that was released when budworm killed the overstory. Dispersed cutting operations in the 1930s and 1940s removed many outbreak survivors. These stands then became similar in structure to those released during the budworm epidemic, further homogenizing the age and stand structure of the eastern spruce-fir forests (Seymour, 1985). Unlike old-growth stands, which are distinctly irregular two-storied stands, these new age classes tended to follow the classic even-aged development pattern quite closely (Seymour, 1985). Given this stand structure, the mean annual basal area increment of one-half to two-thirds of spruce-fir forests would be expected to peak at about the same time, with most stands peaking in the early 1960s (± 10 years) and reducing in increment thereafter.

If the observed post-1960 decline in red spruce growth rate were due to natural maturation alone, the decline should vary by stand, with most but not all stands peaking in the early 1960s. The low-elevation red spruce decline that has been reported to have occurred across this region after the early 1960s may be due to the overall averaging of the data. Differences in growth due to age, elevations, stand density, and stand history, and elevation can be obscured by an overall averaging. For low elevations in northern Maine, Reams and Huso (in press) reported on a stand history effect. Using Hornbeck and Smith's (1985) data for four dominant or codominant red spruce trees at each of 182 plots from northern Maine, they identified two disturbances over that region: one release due to a spruce budworm epidemic which occurred around 1920 (± 5 years) was found for 76% of the plots, and another release centering in the 1940s (1935-1955) was found in 54% of the plots. The plots were classified into four types: no release, 1920s release only, 1940s release only, or both a 1920s and 1940s release. A site was categorized as showing a release if at least two trees showed such a trend. Sites showing no release or only a 1920s release exhibited a steady decrease in radial growth since about 1930, including the period in or after the 1960s. Red spruce released either in the 1940s only or in the 1920s and 1940s showed large growth increases that peaked just after 1960; growth rate then decreased and is now approaching the growth rates of unreleased trees (see Figure 8). This behavior was evident for trees of all age classes.

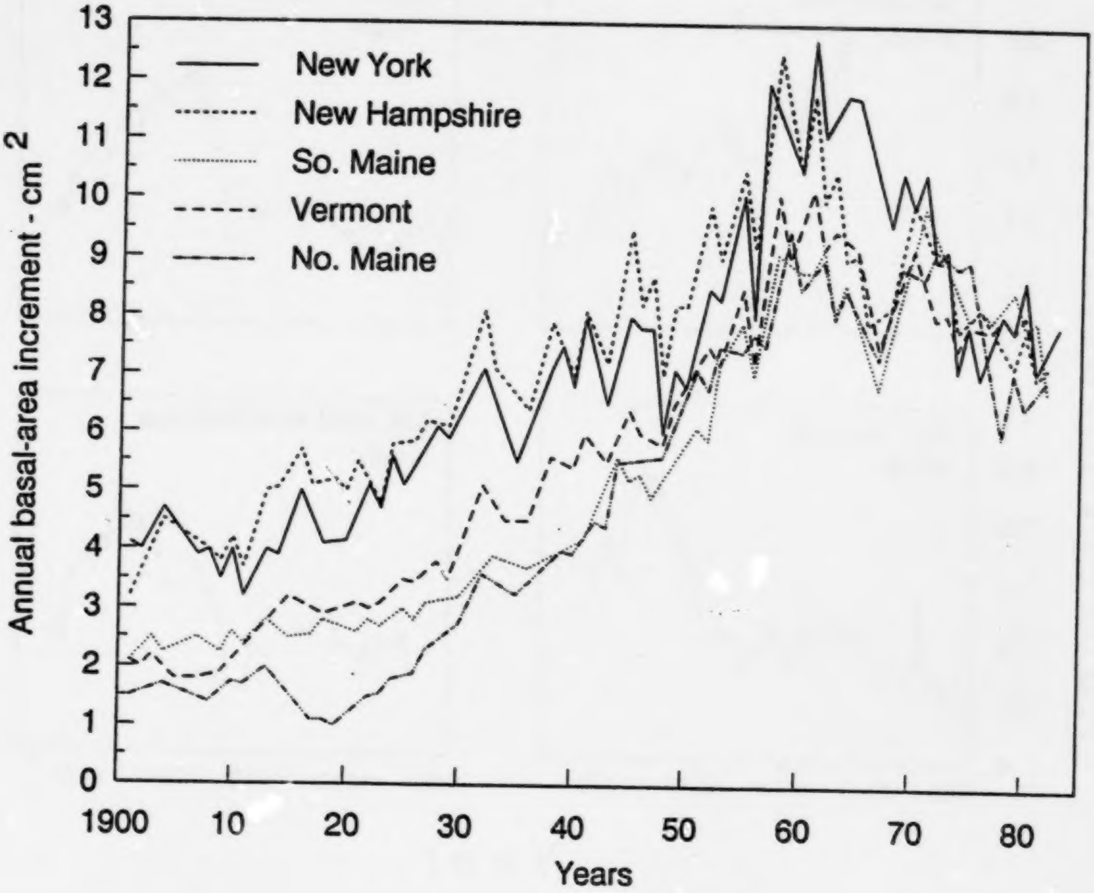

Figure 7. Mean annual basal area increment of red spruce by geographic region and calendar year (from Hornbeck and Smith, 1985).

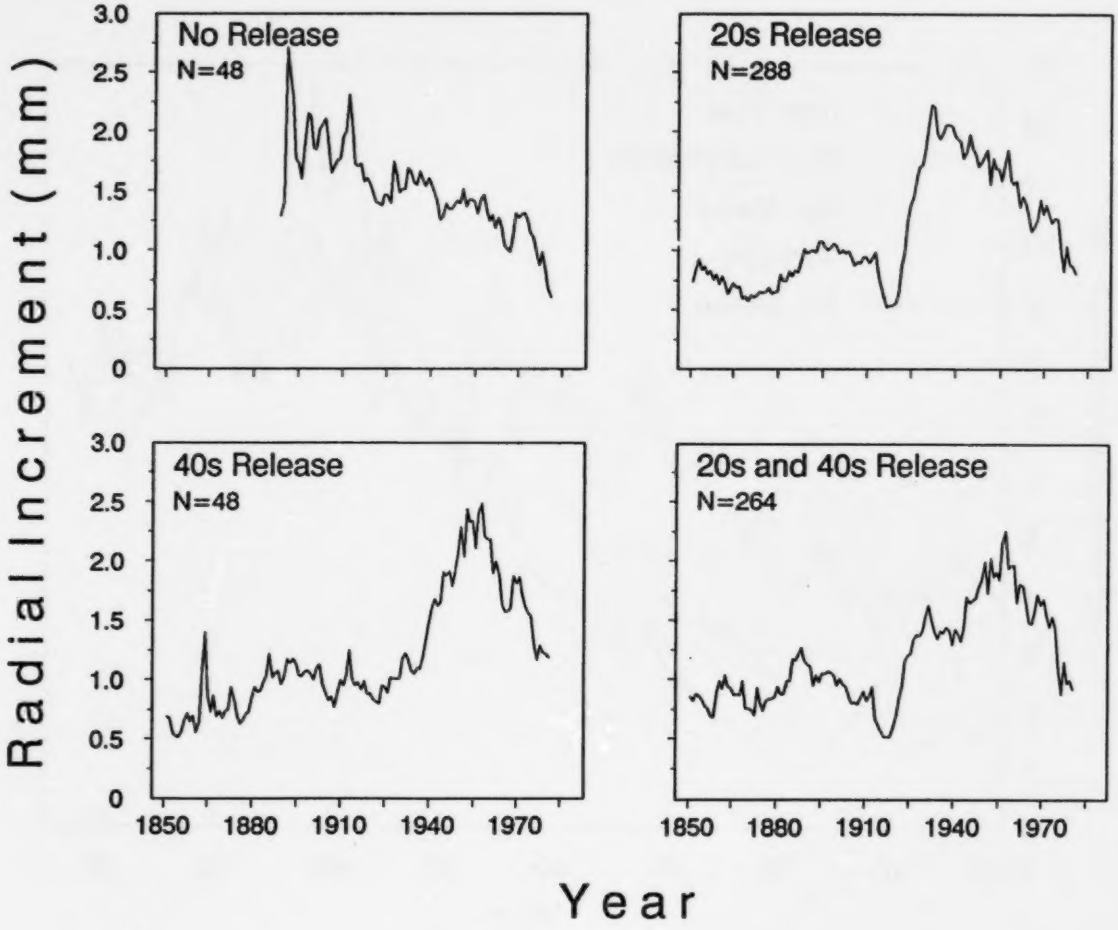

Figure 8. Mean radial increment of red spruce in northern Maine by release history and calendar year (from Reams and Huso, in press).

Reams and Huso (in press) suggested that the increased red spruce growth at low elevations in Maine was related to widespread releases from competition. The causes of these releases are unknown, but the authors suggested that the cumulative coincidence of three factors might be responsible: birch dieback from 1935-1955; beech bark disease, which was epidemic at the same time; and logging, which has continually released trees and which is not solely restricted to the 1935-1955 period.

Van Deusen (in press) reviewed high-elevation transect data of Johnson and Siccama (1983) from Mt. Mansfield, NH, and Whiteface Mountain, NY. Visual inspection of radial increments from individual trees indicated two distinct classes of trees: 50% of trees that were less than 100 years old since release from the understory, and 50% of trees that were over 100 years old since release. The mean site chronology showed a continuous and smooth growth increase prior to 1960, followed by a large reduction in radial increment. He found that for individual cores, trees less than 100 years old declined in radial increment, but that this reduction occurred over a 20-year period starting in 1955. Trees over 100 years old have shown both increases and decreases since 1960. Van Deusen (in press) concluded that the degree of post-1960 growth reduction is related to the proportion of both younger and older trees showing a pre-1950 growth increase (see Figure 9).

Van Deusen (1987b) analyzed both the low-elevation FIA data of Hornbeck and Smith (1985), of which less than 2% were over 100 years old, and 10 additional data sets from stands with trees over 100 years old. The data sets for trees over 100 years came from both high- and low-elevation sites in Maine, New Hampshire, and New York. All data were from Cook (personal communication, Lamont Doherty Geological Observatory), with the exception of Elephant Mountain, ME, data from Conkey (1986a,b). Looking at the time-varying slope and intercepts of linear equations, Van Deusen reported that young red spruce trees (less than 100 years old) had a tendency towards consistently decreased relative growth rates after about 1960, whereas the older trees (over 100 years old) exhibited no distinct trends in any direction. He concluded that older trees are not in general decline and that growth rates of younger trees are decreasing and converging to those for older trees.

Van Deusen (1987b) also reported that the data at high elevations did not show the decline in high-elevation red spruce suggested by some other studies (e.g., Johnson et al., 1988; McLaughlin et al., 1987). One possible explanation for this difference is that Van Deusen standardized by using the logarithm of the first differences, a technique which tends to dampen growth trends. Another reason for the differences may be that the other studies included both old and young trees in the analysis, but Van Deusen's data included only old trees.

Conkey (1989) and Van Deusen (in press) have suggested that the recent high-elevation declines in red spruce radial increment at Mt. Moosilauke and Whiteface Mountain, respectively, were accentuated by nearly synchronous increases in tree growth in the 1940s and 1950s. This increase and subsequent decrease in growth supports a stand dynamics effect at these high-elevation sites, previously suggested by Federer and Hornbeck (1987a,b) for low elevations and by Van Deusen (1987b) for trees at all elevations.

LeBlanc and Raynal (submitted) sampled 46 dominant and codominant crown class red spruce trees from sites at 900 and 1100 m on Whiteface Mountain, NY. The age since release of these trees ranged from 39 to 290 years, dbh ranged from 21 to 52 cm, and height ranged from 13 to 22 m. The sample included equal numbers of apparently vigorous trees, declining trees, and almost dead/recently dead trees. These vigor ratings were based on extent of fine-twig dieback. Vigorous trees had full crowns and less than 25% fine-twig dieback, and poor-vigor trees either were dead or exhibited greater than 90% loss of crown. Each site was selected based on evidence of high red spruce density, the absence of any historical evidence of logging, and absence of

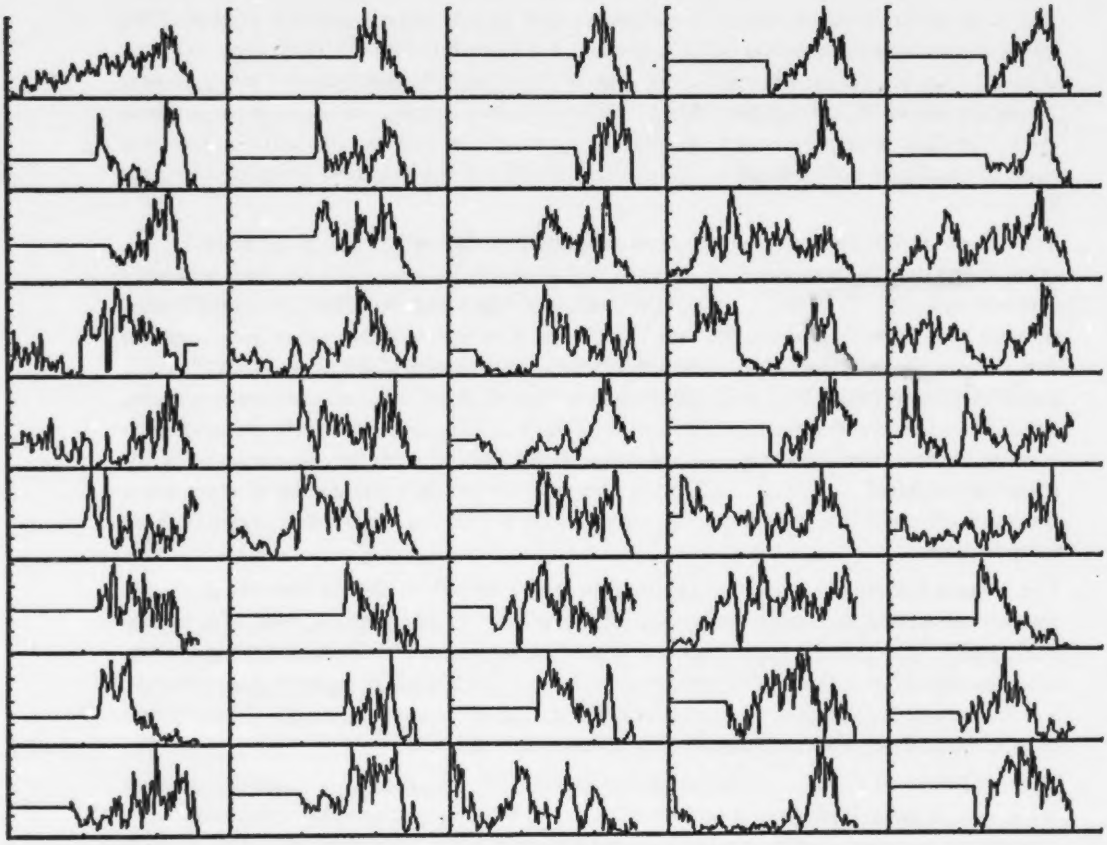

Figure 9. Raw ring widths of red spruce trees from Whiteface Mountain, NY. The upper left corner shows the average chronology and the remaining plots are of individual cores. Missing values are replaced by a straight line at the mean of the series (from Van Deusen, in press).

fir-wave disturbance (wind-induced, cyclic waves of death, regeneration, and maturation). At sites at both elevations, more than 70% of overstory red spruce was standing dead (LeBlanc et al., 1987).

LeBlanc and Raynal (submitted) calculated apical and radial growth indices from internode length and annual ring width. Apical and radial growth declined across both sites. However, the timing of growth reductions differed between young trees (less than 100 years since release) and old trees (more than 100 years since release). For the older trees, reduced mean apical and basal area growth began in the mid-1960s. For young trees at 1100 m, basal area growth declined in 1965, but reduced height growth began in the early 1970s. At 900 m, reduced mean basal area growth of young trees began in 1975, several years after a reduction in height increment.

LeBlanc and Raynal reported that the incidence of terminal leader injury varied among sites and age groups. However, the incidence of injury after 1970 was unprecedented over the last 100 years. Trees at 1100 m had a greater incidence of terminal leader injury than those at 900 m. Incidence of leader injury in young trees from the 1100-m site was similar to that of old trees; however, at 900 m, young trees showed no terminal injury, but apical growth reductions after 1973 may have involved leader injury and recovery. Dead leaders were found on 71% of old trees and 63% of young trees at 1100 m, and 50% of old trees at 900 m. LeBlanc and Raynal (submitted) suggested that leader injury was abnormally high for young and old at both elevations. LeBlanc and Raynal combined the data from both elevations and correlated mean annual stem-wood volume increment from 1945-1954, 1955-1964, 1965-1974, and 1975-1984 with competitor tree basal areas. Mean annual stem-wood volume increment (AVI) was negatively correlated both with competitor tree basal areas from 1945 to 1964 and with the sum of the ratios of competitor tree basal area to sample tree basal area until the last period (1975-1984).

Since 1964, mean AVI was not correlated either with the competitor tree basal areas or with the ratios of competitor tree basal area to sample tree basal area since 1975. The authors suggested that this result indicates a breakdown of the normal relationship between growth and competition. However, recent work by LeBlanc (personal communication) indicates that this relationship only breaks down at 900 m, while at 1100 m mean AVI is negatively correlated with the competition index. Whether these findings indicate a breakdown of the normal relationships at 900 m or 1100 m is not known. The authors suggested that trees with the most competition showed the least growth decline and that emergent trees showed the greatest growth decline. They concluded that coincident apical and radial growth decline for both age groups and sites, repeated leader injury and recovery, and a relationship between crown exposure and decline suggested that climate as well as acid mist and ozone were potential causal factors. A graph of LeBlanc and Raynal's (submitted) data showing mean AVI against age since release shows that 9 of 23 trees at 900 m are growing at or above expected rates, and that growth rates for most trees at 1100 m are below expectation (see Figure 10).

Solomon et al. (in preparation) used data from seven studies of low-elevation red spruce to test for spatial and temporal growth rates that were different from rates expected based on stand dynamics. The data were individual tree and plot remeasurements from 1955-1974 on 774 growth study plots and 311 FIA plots. The plots used were representative of commercial spruce-fir and northern hardwood types in the Northeast (from eastern New York to Nova Scotia, Canada). Three different periods were identified for the temporal analysis: 1955-1964 (period 1), 1960-1969 (period 2), and 1965-1974 (period 3). Since period 2 overlaps with both periods 1 and 3, only comparisons between periods 1 and 3 will be presented here. Solomon et al. used analysis of covariance to investigate whether changes in tree and stand characteristics (tree diameter, stand basal area, quadratic mean stand diameter, and percent hardwoods) were associated with changes in diameter growth. Changes in these variables are assumed to reflect stand dynamics.

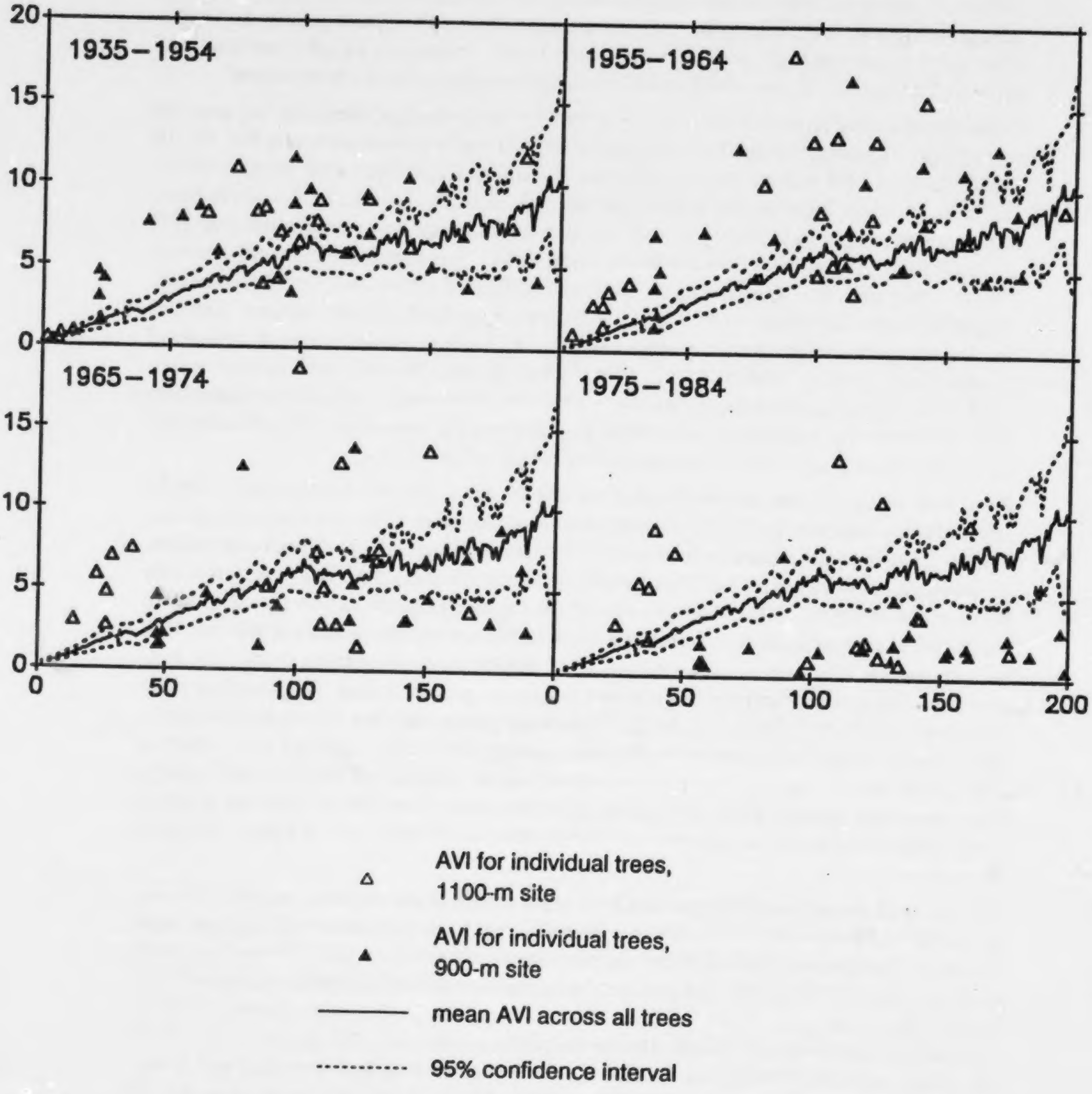

Figure 10. Individual tree periodic mean AVI compared with expected AVI versus age curve (from Leblanc, submitted).

The statistical models accounted for 17% of the diameter growth variation in period 1 and 18% in period 3. Red spruce diameter growth was negatively correlated with basal area, quadratic mean stand diameter, and initial tree diameter. Red spruce mean diameter growth was greater in period 1 than in period 3 ($p = 0.01$), but the difference between the two periods was close to zero ($g_1 - g_3 = 0.022$). After adjusting for the covariates, growth in period 3 was greater than in period 1, leading the authors to conclude that differences in unadjusted means were attributable to stand dynamics.

The effect of climate on growth rates has also been studied. McLaughlin et al. (1987) reported the results of red spruce tree-ring data from a total of 18 stands located at elevations from 100 m to 1160 m on six mountains. The majority of stands were on Whiteface Mountain, NY (7), and Mt. Washington, NH (6). The rest were at Mt. Desert, ME (2), Hubbard Brook, NH, Camels Hump, VT, and Hunter Mountain, NY. Two cores were drawn from opposite sides of each tree and then crossdated. Ring-width sequences for each tree were standardized to remove low-frequency trends in growth that were not well replicated across a stand. To standardize, a negative exponential curve was used, and if this curve did not fit, simple linear regression was applied. Mean chronologies were produced both for each stand and across stands, providing a record of approximately 200 years of spruce growth for the 18 stands. Radial growth peaked around 1955-1960, and the overall chronology showed a downward trend in mean ring-width index beginning about 1960. Equally low indices occurred from 1788 to 1800 and 1817 to 1830. The earlier decreases lasted 10 to 15 years, but the recent trend has shown no reversal after more than 20 years. Extensive red spruce mortality in the 1880s was not reflected in the overall chronology. McLaughlin et al. found no large increase or decrease in radial increment during that period. They concluded that during the past two decades red spruce has responded to factors other than climate, or to climate in an unusual way, and that these deviations may be related in part to atmospheric deposition of pollutants.

McLaughlin et al. (1987) also did an intervention analysis to detect both positive and negative changes in growth that appeared unusual compared with the rest of the growth series. They observed three trends: 1) a strong upswing in positive steps beginning around 1885 and peaking during 1900-1904; 2) three previous upswings in negative steps (1905, 1915, and 1935); and 3) the largest increase in negative step interventions occurring between 1955 and 1960 (see Figure 11). Figure 11 also indicates that the upswing in positive steps from 1920 to 1955 is as great as the 1885-1905 change, and that the increase in negative steps through 1960 appears to follow the increase in positive steps up through 1955. For example, there were approximately 131 positive interventions and 148 negative interventions from 1930 to 1950.

Johnson et al. (1988) obtained increment cores from red spruce trees in 1982 from surveys in the Catskill, Adirondack, Green, and White Mountains. At all sites, tree age ranged from 80 to 230 years. A few sites had been logged around the turn of the century, but most had not been disturbed by logging or fire for 130 years. The cores were cross-dated, and each tree ring-width series was standardized by a tight-fitting smoothing spline that removed all low and intermediate frequency variation. Mean monthly temperatures were correlated with high-frequency variation in ring width (i.e., with standardized ring indices). For high elevations (800 to 1220 m), July and August temperatures of the year preceding ring formation were negatively correlated with the standardized ring indices. December and January temperatures were positively correlated with the ring indices. The climate-high frequency growth model was calibrated for 1885 to 1940. Comparisons of actual versus predicted standardized ring widths indicated that a change in the climate-growth relationship began in the early 1960s (see Figure 12). The authors suggested that climatic stress and pollutant exposures are possible causes of red spruce decline.

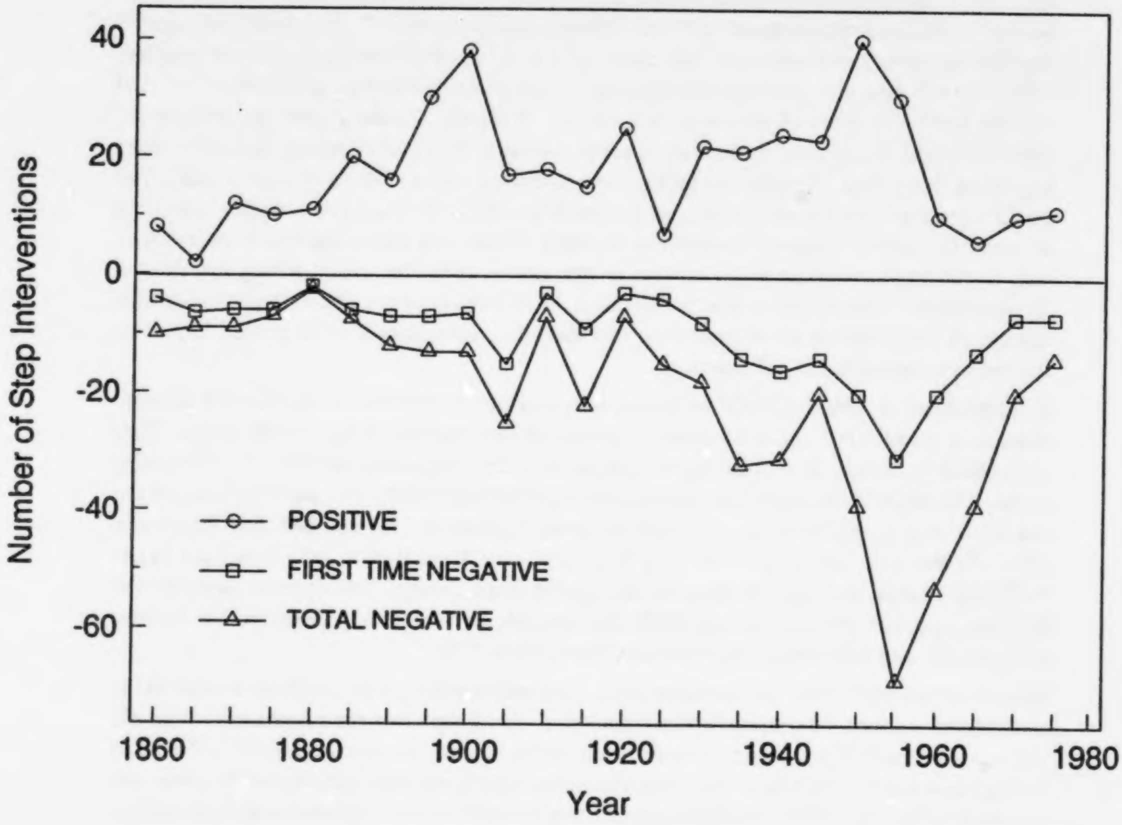

Figure 11. Frequency of step interventions in five-year intervals (redrawn from McLaughlin et al., 1987).

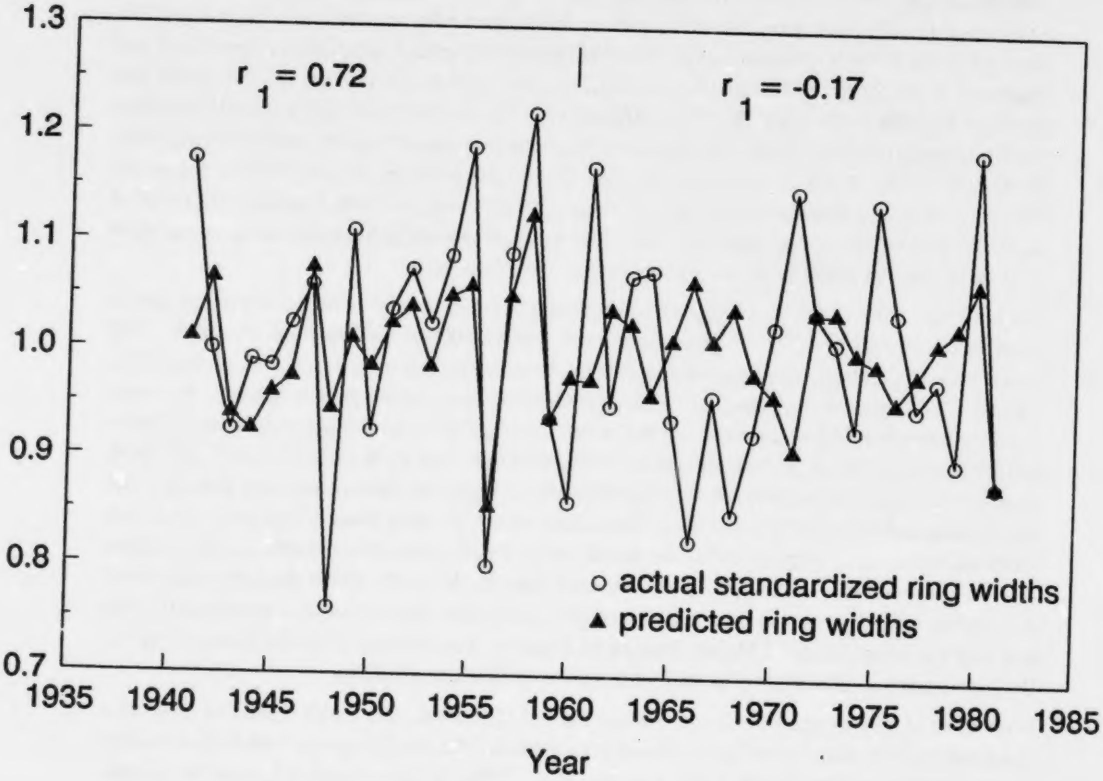

Figure 12. Actual standardized ring widths and ring widths predicted by the August-December temperature model at Whiteface Mountain, NY, Mt. Mansfield, VT, and Mt. Washington, NH. Correlation coefficients are for predicted versus observed ring widths for 1941-1960 and 1961-1981 (redrawn from Johnson et al., 1988).

Cook et al. (1987) summarized and reanalyzed three dendrochronological studies of red spruce: McLaughlin et al., (1987); Cook, (1985b); and Johnson et al., (1988). The study sites sampled and modeled represented 14 sites above 800 m at seven mountains in New York and Vermont. Cook et al.'s methodology was essentially the same as McLaughlin et al.'s (1987). For climate modeling, average temperatures of 17 months (from the May prior to, through September of the year of, ring formation) were candidate predictors. The results were similar to McLaughlin et al.'s: the previous July and August had negative coefficients; November, December, and the current-season May and September had positive coefficients.

Cook et al. (1987) also reported on Johnson et al.'s (1988) study of extremes in August and December temperature sequences for years in which mean temperature deviated from the long-term mean by more than 1.64 times the estimated long-term standard deviation. Cook et al. concluded that there were years of notable temperature stress that were separated by a decade or more of non-stressful years: the 1870s, the late 1930s, the 1940s, and the late 1950s-early 1960s. Each of these periods coincided with abnormally high red spruce mortality in New York and Vermont in the 1870s, 1930s and 1940s, as well as the onset of the current decline in the late 1950s-early 1960s (see Figure 13). The authors stated that the 1870s mortality, the only previous decline thought to be as serious as the present, showed severe August and December stress similar to the late 1950s. Cook et al. concluded that the correspondence of temperature stress and mortality suggested that abnormally high climatic stress may be a predisposing factor or cause of past and present red spruce decline. This does not rule out air pollution as a contributing or inciting factor that might impede recovery at high-elevation sites.

Conkey (1984, 1986a, 1988) investigated forest health using intra-ring wood density chronologies from 26 red spruce trees at a high-elevation site (above 800 m) on Elephant Mountain, ME. Conkey found that both densities and ring widths appeared to decrease after a peak around 1960. The density series looked unusual in the last decades; no earlier period showed the same low-frequency dominated downward trend or low standard deviation. Ring widths, on the other hand, showed a gradual decline similar to earlier declines, such as in 1862-1876 and 1907-1920. Conkey also reported that the relationship between spring temperatures and ring densities was remarkably stable until 1964, but since then there seems to have been a departure from this relationship. Conkey suggested that the density record indicates a stand health problem not yet seen in ring widths. Although reports indicate that in the early 1960s drought and winter desiccation were observed in west and central parts of Maine with subsequent heavy needle loss and bud mortality (State of Maine, Bureau of Forestry, Entomology Division Annual Report, 1962), it is unclear how these climate factors relate to Conkey's findings.

Federer et al. (1989) used the low-elevation data of Hornbeck and Smith (1985) to develop a dendroclimatic model for red spruce basal area growth. The results agreed with high-elevation dendroclimatic studies (Cook, 1987; Johnson et al., 1988) in that detrended diameter growth correlated negatively with previous year summer temperatures (July and August), and positively with early winter temperature. Federer et al. also found an effect of current September temperature; however, they suggested that this effect was spurious because it was due solely to one year (1961) with wide rings and an unusually warm September.

Thus, several studies have suggested that the response of Northeastern red spruce to climate has changed since approximately 1960. Of this research, none of the dendroclimatic models accounted for more than approximately 30% of the variance. Arbaugh and Peterson (1989) reported that, in general, the regression techniques most commonly used in dendroclimatic modeling reliably reflect the correct climate variables only when explained variability was greater than 70%. Reliability of variable selection decreased rapidly as unexplained variability and multicollinearities increased. This finding argues for validation of any model using independent

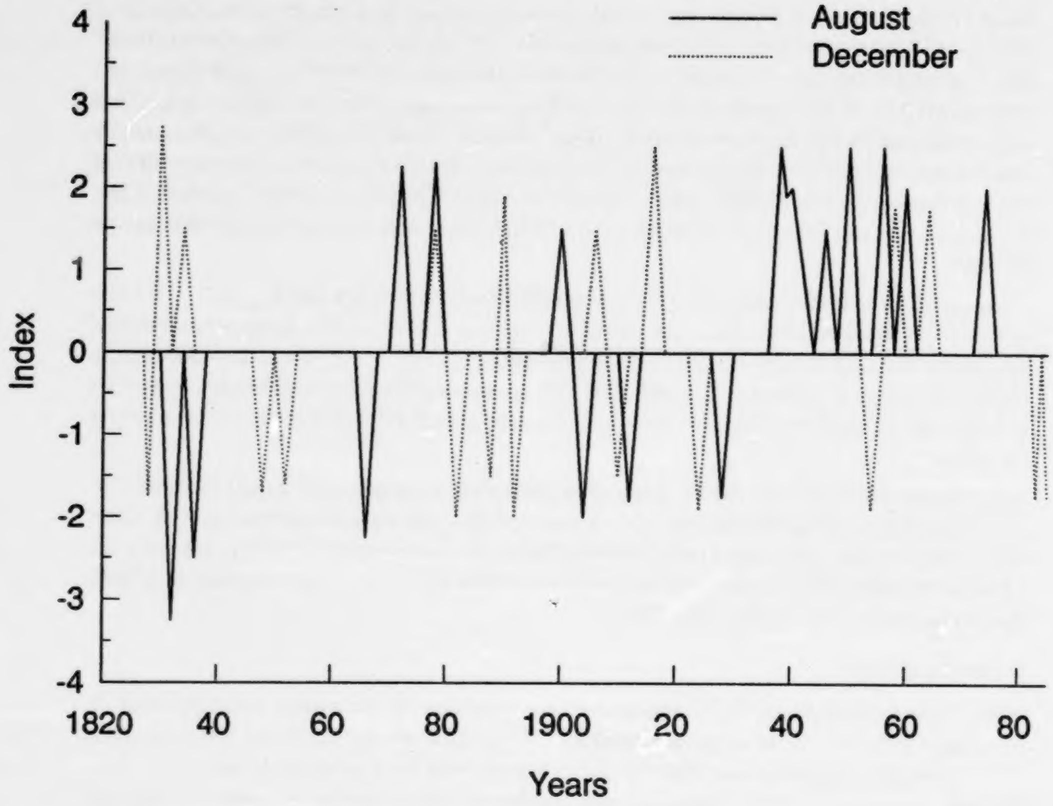

Figure 13. August and December stress indices. Solid and dotted lines represent temperatures exceeding ±1.64 standard deviations for, August and December respectively (redrawn from Cook et al., 1987).

data sources. Fortunately, because different sources have been used to develop northern red spruce dendroclimatic models, that validation has taken place, with summer and early winter temperatures always statistically significant for all studies.

In addition to dendroclimatic research, other surveys of forest growth have been conducted. Barnard and Scott (1988) compared basal area growth rates of 19 tree species for the two periods from 1965-1972 and 1973-1983. The trees were from 435 FIA plots representing a variety of forest conditions in Vermont. The purpose of the study was to identify species that may be experiencing growth reductions. White spruce and red spruce were the only species for which the average annual basal area growth decreased. Barnard and Scott recommended further investigation into the causes of these reductions.

Peart (1989a,b) assessed growth rates of red spruce and balsam fir saplings and seedlings from 1983 to 1987 on the eastern slopes of Mt. Moosilauke, NH. At each of the 12 permanent 400-m^2 plots, all spruce saplings were measured: there were three saplings at 840 m, six at 990 m, and three at 1140 m. If there were more than 50 balsam fir saplings in the plot, balsam fir saplings were measured in one randomly located 100-m^2 subplot. Each live sapling was permanently marked and measured for height growth. Saplings were classified as showing less than 10% of recent foliage loss (class 1), 10% to 50% foliage loss (class 2), or 50% to 100% foliage loss (class 3). Red spruce and balsam fir seedlings (< 1 m tall) were sampled along contours at the same elevations.

Peart reported that red spruce and balsam fir saplings have increased in height growth at the low and middle elevations since 1983. Growth of both species decreased with increasing elevation. Red spruce seedling density was lowest at high elevations (< 100 seedlings/ha). Low and middle elevations had approximately 700 and 2200 red spruce seedlings/ha, respectively. Seedling growth rates of the two species were comparable, and growth rates decreased with increasing elevation.

In a separate study (Peart, 1989c), permanent plots were established in spruce-fir forests on east-facing slopes of Mt. Moosilauke, NH, in 1986, and the plots were remeasured in 1988. Over the two-year period, mortality of red spruce and balsam fir over 5 cm dbh was 2% to 3% per year. Dead trees tended to be those in poorest canopy condition since 1986. Crown conditions of both species have improved slightly since 1986.

Crown Condition

Miller-Weeks and Cooke (1989) surveyed crown condition of red spruce and balsam fir in permanent plots at a wide range of elevations (300 to 1200 m) in New York, Vermont, New Hampshire, Massachusetts, and West Virginia. Surveys have been conducted yearly since 1985. The authors reported a deterioration in crown condition of spruce and fir over time in all regions. Little overall discoloration in the crowns was noted, but foliage loss due to branch mortality or dieback was prevalent. Other crown symptoms noted were broken branches and bunching of lateral shoots following terminal dieback. Needle retention on foliated red spruce branches was good. Chlorotic mottling was common on sampled red spruce branches. Several insects and pathogens, as well as wind, ice, and snow, have been associated with some of these symptoms. However, there was no observed change in the number of recently dead trees. The authors concluded that crowns are deteriorating but the trees are surviving.

Wargo et al. (1989) studied root systems, crown conditions, and radial growth of 21 red spruce trees at 950 m on the western slope of Mt. Abraham, VT. Dominant and codominant crown class trees were selected in a spruce-fir stand exposed to the prevailing southwesterly winds. Nine trees from each of two decline classes were selected (class 1, 0-10% crown deterioration; and class 2, 11-50% crown deterioration). Three trees with severe decline (class 3, 50-100% crown deterioration) were selected in a nearby stand. Slower radial growth since the early 1960s was associated with deterioration of the crowns and root systems. Trees in crown class 1 had greater radial increment than the other two classes ($p < 0.05$), and trees in classes 2 and 3 showed a reduction in radial increment in the early 1960s. The healthy trees (class 1) had more living fine roots and more mycorrhizal tips than the class 2 and 3 trees.

Insects and Pathogens

In a review of historical impacts on northeastern spruce-fir forests, Weiss and Millers (1987) reported that logging, insects, and adverse weather have been important factors in the past 150 years. In mountainous areas, logging and bark beetle outbreaks almost eliminated the original forest, and red spruce did not return to many of the sites it had formerly occupied. Although red spruce did return to upper slopes, the conditions are not optimal for growth. Periodically, the spruce beetle has caused mortality in the few remaining old-growth stands. Four spruce budworm outbreaks, primarily in Maine, caused damage in 1878-1887, 1910-1919, 1945-1955, and 1970-1985. Thousands of acres of balsam fir and spruce in Maine, far northern New Hampshire, and northeastern Vermont died in the last outbreak. Both the spruce budworm and logging have created openings favorable to increases in balsam fir. The authors concluded that red spruce and balsam fir have been affected by winter injury, wind, and possibly drought and that reductions in red spruce vigor in the Northeast have been due at least in part to insects, pathogens, and wind.

Weidensaul et al. (1989) surveyed 60 plots located from 800 to 1100 m on Whiteface Mountain, NY, in the summer of 1987. They looked for visible symptoms of insect and pathogen activity. Weidensaul et al. reported some fungal pathogens (*Uredinopsis* spp.), but they concluded that at the time of the survey these pathogens were not causing significant problems.

Smith (1988) surveyed spruce-fir forests at Mt. Moosilauke, NH, for pathogen signs and symptoms. No agent of major damage to red spruce was identified, but high counts of nematodes (22,000 to 46,000 per m^2) were found. The effects of high nematode densities on red spruce are not clear.

Tobi et al. (1988) observed larvae, pupae, and adults of the swift (ghost) moth at all elevations (700 to 1150 m) in a transect at Camel's Hump, VT. The larvae reportedly feed on fine roots of red spruce and balsam fir. Girdling injury similar to that caused by the moth was evident on a large number of dead seedlings. The authors considered the moth to be a potential factor in the decline of high-elevation balsam fir and red spruce.

Knight and Grosman (1989) observed large numbers of swift (ghost) moth at three elevations (750, 960, and 1110 m) on Mt. Moosilauke, NH, and two study plots at Howland, ME. The spruce budworm and spruce beetle were present at most sampling sites in low numbers. The relative absence of insects or pathogens at epidemic proportions appeared to exclude biotic agents as key factors in mortality of individual trees.

Soils

Soils in northeastern forests have also been studied, and this research is discussed in greater detail in MPO #4 (Mattson et al., 1990). Some typical trends are shown in a study by Johnson et al. (1989). They found decreases in soil calcium (Ca) and magnesium (Mg) and increases in potassium (K) and aluminum (Al) as elevation increased from 700 m to 1300 m on Whiteface Mountain, NY. Foliar Mg content also decreased with elevation, but was not related to crown damage in red spruce.

Similarly, Huntington and Ryan (1988) found that within a given soil type exchangeable Ca and Mg decreased, both in the forest floor and in the mineral soil, as elevation increased from 840 to 1200 m on the eastern slopes of Mt. Moosilauke. As elevation increased, a decrease in K was observed in the forest floor but not in the soil. Concentrations of exchangeable Al increased with elevation, while soil pH decreased with elevation. Available phosphorus (P) increased with elevation for spodosol soils, but decreased in cryofolist soils. Lead (Pb) showed increases in spodosols with elevation, while zinc (Zn) decreased. It is difficult to draw conclusions about soil chemical changes due to deposition from these data, since soil physical and chemical properties normally change as a function of elevation.

Summary

For many high-elevation sites in the Northeast, large proportions of recent standing dead red spruce have been reported since the mid-1960s, and this mortality appears to be related to frequent winter injury events starting in the late 1950s and early 1960s. The proportion of recent dead red spruce is greater in the Adirondack and Green Mountains than in mountains to the east (30% versus 14%). The Adirondack and Green Mountains show increased standing dead red spruce with increasing elevation, while in the eastern mountains standing dead red spruce has not increased with elevation. The greatest mortality has occurred at or above 1000 m, an elevation that may represent an extreme in the range of red spruce. Thus, an environment that may already be considered marginal for red spruce may become intolerable for the species if additional stresses are imposed, such as an unfavorable climate and/or acidic deposition. A regional survey of red spruce crown condition found that since 1985 crown condition has deteriorated, but no change was observed in the number of recently dead trees.

Red spruce radial growth data indicate a widespread pre-1950 radial growth increase and a subsequent post-1960 growth decrease at low and high elevations throughout the northeastern United States. The percentage of sites in the region that show the 1960s growth reduction is unknown because data were aggregated over sites and because studies were not specifically designed to determine this percentage. One study restricted to northern Maine suggests that approximately 54% of low-elevation sites sampled show a pre-1950s increase followed by a post-1960s decrease. Research at high elevations suggests that closely spaced years of winter injury in the late 1950s and early 1960s which resulted in loss of current-year needles could be the cause of radial growth reductions and a breakdown of the high-frequency growth-climate relationship since the early 1960s. Other research suggests that stand history events could explain the apparently synchronous changes in radial growth at both high and low elevations, independent of a regional climate signal found in all age classes. To test these various hypotheses, future work should correlate changes of radial increment with winter injury and stand history for individual trees at a number of sites.

4.1.2 Wet Deposition

Spatial variation in the 1985-1987 precipitation deposition of H^+, SO_4^{2-}, NO_3^-, and NH_4^+ in the Northeast is presented in Figure 14. The Northeast generally received the highest deposition of all forested areas examined across the United States. Within the Northeast, deposition of all four ions was higher in New York, Ohio, and Pennsylvania than in Maine. For precipitation deposition, H^+ and SO_4^{2-} ions generally exceeded 500 eq/ha/yr in the first three states, while northern Maine received less than 300 eq/ha/yr, and central/southern Maine received about 300 to 400 eq/ha/yr. For northeast sites above 250 m, SO_4^{2-} deposition via precipitation can exceed 600 eq/ha/yr (Mohnen, 1988a).

In the Northeast, NO_3^- deposition from precipitation was typically about 50% of SO_4^{2-} deposition, while NH_4^+ was about 25% of H^+ deposition. Precipitation concentrations were typically from pH 4.3 to 4.7, with 1 to 3 mg/L SO_4^{2-}. Figure 14 suggests that the variability in wet deposition between nearby sites was considerable, and that differences existed between Maine and the states closer to the Ohio Valley (New York, Pennsylvania, and Ohio).

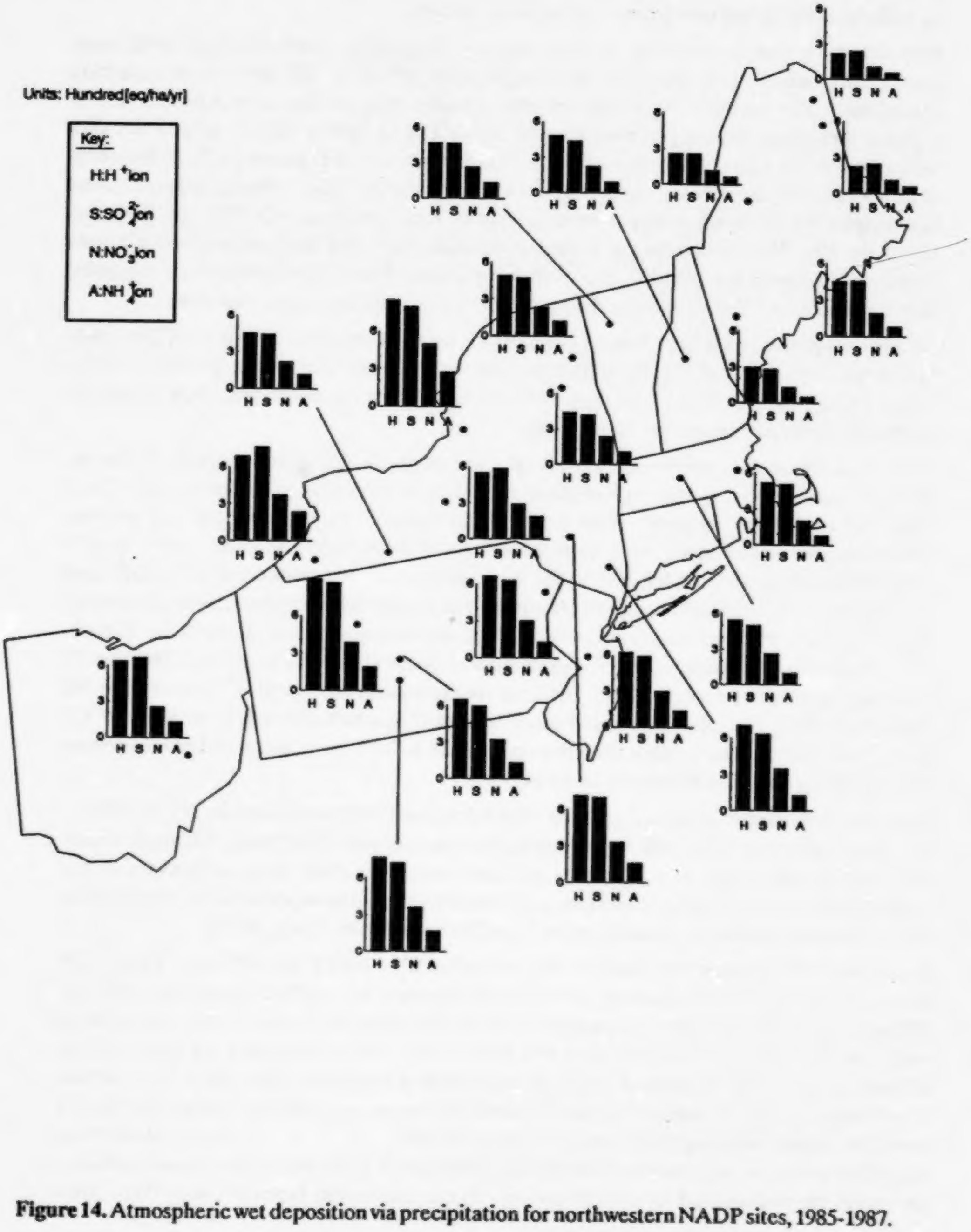

Figure 14. Atmospheric wet deposition via precipitation for northwestern NADP sites, 1985-1987.

With respect to elevational variation in acidity, cloud water deposition of H^+, SO_4^{2-}, NH_4^+, and NO_3^- represented a significant input to forest canopies in the northern and southern Appalachian Mountains. Cloud water deposition often exceeded the flux from precipitation during the growing season at mountain sites frequently exposed to cloud; total deposition above cloud base (about 1000 m) substantially exceeded deposition below cloud base. Cloud water concentrations of H^+, SO_4^{2-}, NO_3^-, and NH_4^+ were much greater than precipitation concentrations (see Figure 15). An elevational difference in acidity results, since only precipitation occurs below about 1000 m, while both cloud and precipitation occur above 1000 m.

Elevational differences also exist in cloud exposure frequency. Regional cloud climatology studies have indicated typical cloud base heights from 800 m to 1200 m in the Appalachian Mountains. The northern Appalachians were cloudier than the southern Appalachians in regional climatology studies performed for the MCCP (Vong, 1989a). MCCP data for five sites indicated that the Appalachian peaks (above 1400 m) were in cloud approximately 28% to 42% of all hours during the growing season (April-October). Figure 16 shows the frequency of cloud base heights for hours when clouds were present at Whiteface Mountain, NY, and Whitetop Mountain, VA. Year-round data at Whitetop Mountain show that the peak was in cloud more frequently during the non-growing than the growing season. Winter data on cloudiness at specific sites are scarce such that growing and non-growing season differences are uncertain.

Clouds were present on at least one out of three days for elevations from 1000 m to 1400 m, while sites above 1400 m had clouds up to 75% of days during the growing season (Mohnen, 1988b; Vong, 1989a). These cloud base data indicate that higher elevations are likely to receive substantial chemical inputs from cloud water.

Elevational differences in chemical exposure also were seen. In high-elevation spruce-fir forests, observed cloud water pH was approximately 0.6 pH units lower than precipitation pH. Cloud water pH of 3.0 was frequent, while a pH of 2.8 occurred during most growing seasons. Concentrations of the major ions were from 5 to 20 times higher in cloud water than in precipitation, depending on location in the Appalachians. Concentrations of H^+, SO_4^{2-}, and NO_3^- greater than 100 μeq/L occurred infrequently in precipitation but routinely in cloud water (more than 40% of cloud hours) at the five MCCP monitoring sites and at two of the Electric Power Research Institute's (EPRI) Integrated Forest Study (IFS) sites in the Appalachians (S. Lindberg, personal communication). Extreme concentrations (H^+ or SO_4^{2-} greater than 500 ueq/L, and NO_3^- greater than 250 ueq/L) occurred from 12% to 42% of cloudy hours at the MCCP sites. Thus, below-cloud eastern forests were exposed to both lower mean and fewer extreme concentrations than sites frequently in cloud.

There was a significant altitudinal gradient within clouds at Whiteface Mountain, NY, for NH_4^+, H^+, SO_4^{2-}, and NO_3^- ions, with higher concentrations at or near cloud base. Although within-cloud elevational trends in acidity have not been studied at other sites, an increase in the concentrations of wet deposited ions (i.e., a pH decrease) would be expected in the Appalachian Mountains as elevation decreased to cloud base (Mohnen, 1988b; Vong, 1989a).

Elevational differences in wet chemical flux are indicated by MCCP and IFS data. The MCCP has monitored cloud water chemistry at Whiteface Mountain, NY, and Mt. Moosilauke, NH, and IFS has collected cloud water at a lower elevation on Whiteface Mountain. Comparison of cloud water chemical deposition at these sites with four sites in the Southeast does not show regional differences, primarily because of the large uncertainties associated with cloud water inputs. Cloud water deposition could be expected to increase with elevation above cloud base for specific areas, but, again, modeling uncertainties preclude definite conclusions. A review of cloud water deposition estimates suggests that published growing season cloud water interception estimates (in cm/yr) are proportional to cloud frequency, with an uncertainty factor of two to three when

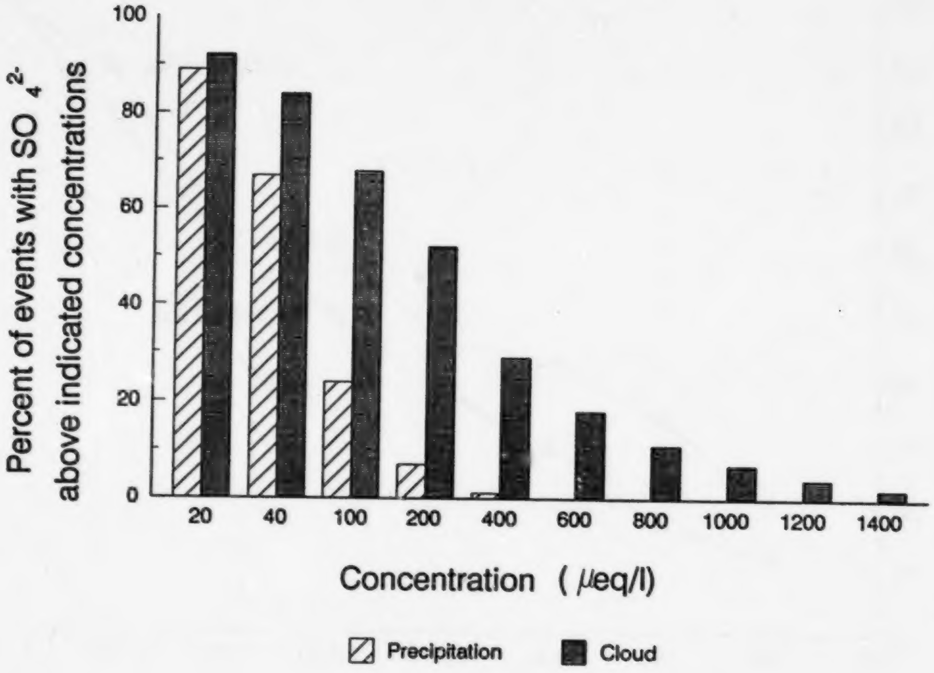

Figure 15. Comparison of SO_4^{2-} distributions for cloud and rain water. Values are percent of cloud samples at Whiteface MCCP site and percent of precipitation samples at Whiteface MAP3S site.

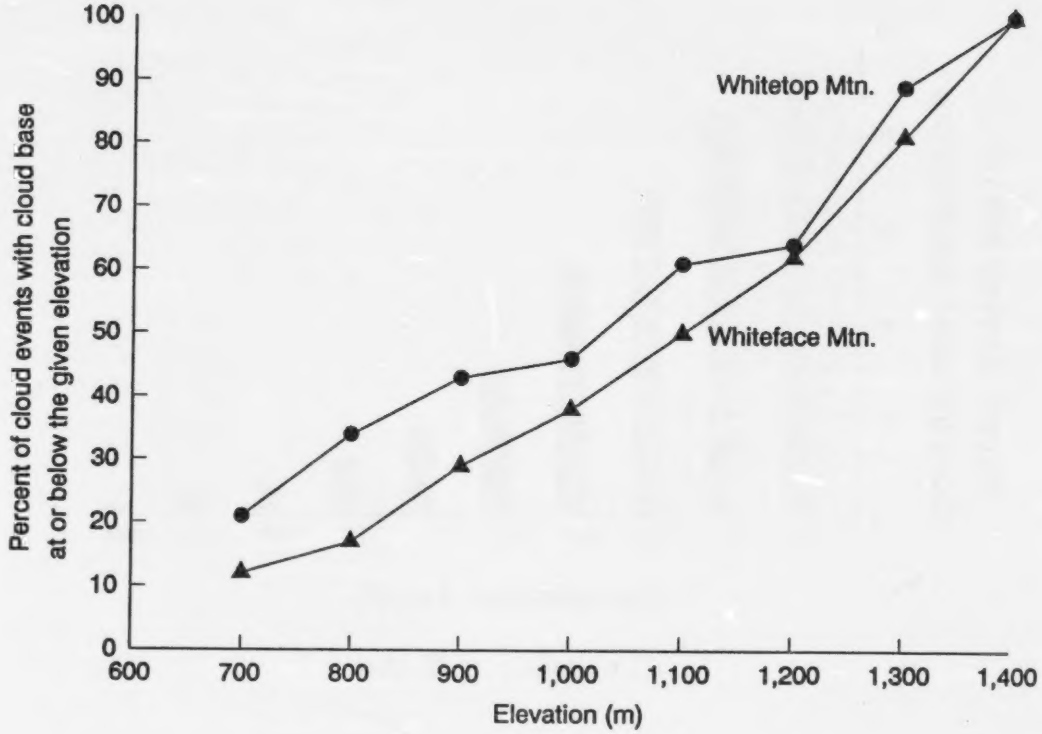

Figure 16. Cloud frequency by elevation during growing season for Whiteface Mountain, NY, and Whitetop Mountain, VA. Total cloud impaction frequencies at the summits are 28% and 42% of all hours for Whitetop and Whiteface, respectively (redrawn from Vong, 1989a).

different sites are compared (Vong, 1989b). When this uncertainty in water interception is combined with a factor of two uncertainty in mean cloud water concentration at a given site (due to height above cloud base), an overall uncertainty of about a factor of six exists in regional cloud water chemical deposition gradients (i.e., New York to New Hampshire).

Whiteface Mountain, NY, (elevation 1483 m) had an estimated warm-season cloud water SO_4^{2-} deposition that was higher than measured precipitation deposition by a factor of two to five, depending upon modeling assumptions. Whitetop Mountain, VA, (elevation 1686 m) exhibited equally high cloud SO_4^{2-} deposition. Mt. Mitchell, NC, (elevation 1950 m) had an estimated cloud water SO_4^{2-} deposition about three times higher than precipitation deposition. The combined NH_4^+ and NO_3^- deposition from cloud interception also was high for the high-elevation MCCP sites at Whiteface Mountain, Whitetop Mountain, and Mt. Mitchell. For the other MCCP sites (Shenandoah and Mt. Moosilauke, elevation 1015 m and 1000 m, respectively), precipitation deposition equaled or exceeded cloud deposition flux for NH_4^+, SO_4^{2-}, and NO_3^-, primarily because of lower cloud frequency (Mohnen, 1988a; Vong, 1989a).

Cloud interception, therefore, has the potential to be a primary source of sulfur (S) and nitrogen (N) deposition to montane forests. For MCCP and IFS sites above 1200 m, cloud water deposition contributed more S and N to forests than did wet and dry deposition. However, there may be substantial uncertainties in deposition estimation procedures for cloud water interception in mountain forests.

4.1.3 Ozone

Sites in the eastern United States for which O_3 concentrations were assessed were identified using the FRP's Atmospheric Exposure Cooperative's Data Book and the AIRS database. Site selection favored remote and forested sites. However, in most cases sites were rural/agricultural or within urban areas (see Table 3 and Figure 2).

In the Northeast, seven sites representative of north-south gradients along the coast and inland, and of elevational gradients are discussed here: Hancock, ME; Whiteface Mountain, NY; Sudbury, MA; Hampden, MA; Wayne, NY; Scranton, PA; and New Brighton, PA. The results in Table 4 indicate that:

1) high-elevation sites such as Whiteface Mountain were exposed to median O_3 concentrations (O_3^{med}) between 40 and 60 ppb, especially during May through August. During extreme episodes (O_3^{worst}), levels as high as 200 ppb have been recorded.
2) O_3^{med} concentrations across low-elevation, non-urban sites averaged between 30 and 50 ppb. Forests near these sites were exposed to concentrations exceeding 80 ppb in several instances, and in excess of 120 ppb on occasion. Concentrations were elevated during nighttime hours, usually by approximately 20 to 30 ppb, although at Whiteface Mountain, nighttime concentrations could reach 60 to 100 ppb.
3) Areas most often exposed to concentrations greater than 80 ppb during the growing season were at higher elevations in the Appalachians, at Hampden and Sudbury. In general, the highest peak concentrations were at sites near urban centers such as Sudbury, Wayne, and possibly Hampden.

In conclusion, high-elevation forests in mountains of northeastern states experienced median concentrations between 40 to 60 ppb. Diurnal fluctuations in hourly O_3 concentrations were suppressed, although on occasion hourly concentrations in excess of 100 ppb were recorded.

Table 3. Site information for AIRS ozone sites in the eastern United States. Numbers refer to Figure 2.

Number	Site Name	Latitude	Longitude	Elevation (msl)	AIRS Description	summary period
1	Sherburne, MN	45° 25.8' N	93° 44.2' W	301	rural-industrial	Jan 80-Oct 82
2	Dickinson, MN	45° 47.1' N	87° 57.0' W	317	rural-industrial	Jan 85-Dec 87
3	Washington, MN	45° 07.7' N	92° 54.1' W	303	city center-industrial	Mar 80-Oct 87
4	Buffalo, WI	44° 16.9' N	91° 51.2' W	388	city center-commercial	Jan 81-Oct 81
5	Marathon, WI	44° 50.7' N	89° 34.2' W	631	suburban-residential	Apr 84-Oct 87
6	Juneau, WI	44° 03.1' N	90° 03.5' W	276	rural-industrial	Jan 79-Sep 79
7	Wayne, NY	43° 13.8' N	77° 10.4' W	140	city center-commercial	Jan 80-Oct 87
8	Whiteface, NY	44° 23.4' N	73° 54.6' W			Apr 86-Dec 87
9	Hancock, ME	44° 39.1' N	68° 36.7' W	213	suburban-industrial	Apr 85-Oct 87
10	Hampden, MA	42° 03.7' N	72° 40.7' W	98	rural-agricultural	Apr 83-Oct 87
11	Sudbury, MA	42° 23.1' N	73° 21.3' W	40	city center-commercial	Apr 80-Oct 87
12	N Little Rock, AR	34° 49.8' N	92° 15.6' W	164	rural-agricultural	Jan 79-Dec 87
13	New Brighton, PA	40° 41.1' N	80° 21.6' W	1300	suburban-residential	Jan 87-Dec 87
14	Jefferson, AL	33° 42.3' N	86° 40.1' W	201	city center-commercial	Jan 80-Dec 87
15	Montgomery, AL	31° 58.3' N	86° 08.1' W	433	rural-agricultural	Mar 86-Nov 87
16	Muscogee, GA	32° 32.4' N	84° 50.6' W	122	rural-agricultural	Jan 80-Dec 87
17	Rockdale, GA	33° 35.1' N	84° 04.0' W	219	rural-agricultural	Jan 79-Dec 87
18	Greenville, SC	34° 54.1' N	82° 14.3' W	287	suburban-commercial	Jan 79-Dec 87
19	Union, SC	34° 32.4' N	81° 33.7' W		suburban-residential	Jan 87-Dec 87
20	Mt Pleasant, SC	32° 56.5' N	79° 39.5' W	5	rural-agricultural	Mar 87-Dec 87
21	Mecklenburg, NC	35° 20.9' N	80° 41.6' W		city center-residential	Jan 80-Dec 87
22	Charlotte, NC	35° 17.1' N	80° 40.2' W	610	city center-commercial	Feb 79-Sep 79
23	Cumberland, NC	35° 05.6' N	78° 47.6' W	40	city center-industrial	Jan 81-Dec 87
24	Davie, NC	35° 52.2' N	80° 25.8' W	244	rural-agricultural	Mar 82-Nov 85
25	Person, NC	36° 17.0' N	79° 07.6' W	198	rural-agricultural	Mar 82-Nov 86
26	Wake Forest, NC	35° 58.3' N	78° 29.4' W	87	suburban-residential	Mar 80-Oct 87
27	Johnston, NC	35° 34.3' N	78° 11.3' W	52	rural-agricultural	Apr 83-Nov 86
28	Edgecomb, NC	35° 59.0' N	77° 35.1' W	18	rural-agricultural	Mar 83-Oct 87
29	Martin, NC	35° 43.5' N	76° 59.6' W	12	rural-industrial	Mar 83-Nov 86
30	Hanover, VA	37° 38.2' N	77° 19.0' W	59	suburban-commercial	Jan 80-Dec 87
31	N Hampton, VA	37° 17.4' N	75° 58.3' W	12	suburban-commercial	Mar 80-Oct 87
32	Dickey Ridge, VA	38° 52.0' N	78° 12.0' W			Jan 83-Dec 87
33	Scranton, PA	41° 35.5' N	76° 04.0' W			Jan 79-Dec 79
34	Indian River, DE	38° 34.6' N	75° 14.8' W			Jan 79-Dec 79
35	Duncan Falls, OH	39° 51.0' N	81° 53.1' W			Jan 79-Dec 79
36	Rockport, IN	37° 52.6' N	87° 03.5' W			Jan 79-Dec 79
37	Lewisburg, WV	37° 46.5' N	80° 20.0' W			Jan 79-Dec 79

Table 4. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the seven AIRS sites in the northeastern United States (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Hancock	Whiteface	Sudbury	Hampden	Wayne	Scranton	New Brighton
May	37-48 (92)	47-56 (96)	20-43 (95)	18-51 (110)	24-48 (117)	20-59 (101)	14-52 (84)
June	33-43 (83)	46-58 (479)	17-49 (162)	18-49 (122)	23-52 (117)	19-68 (103)	22-56 (93)
July	28-38 (123)	42-48 (90)	13-58 (132)	14-55 (185)	22-52 (116)	16-74 (120)	16-54 (90)
August	26-35 (105)	40-49 (91)	9-53 (147)	8-48 (145)	19-47 (104)	22-59 (97)	16-56 (90)
September	24-32 (78)	34-39 (78)	8-40 (123)	6-38 (100)	14-39 (105)	9-49 (123)	8-37 (68)
October	29-34 (57)	33-39 (144)	7-33 (84)	7-32 (61)	13-30 (77)	21-37 (70)	11-30 (50)
% hours (May-Oct) ≥ 80 ppb	1.5	3.5	3.5	3.3	2.7	7.1	2.2

4.2 Southern Forests

Spruce-fir forests occur on 26,610 ha of island-like areas at high elevations in the southern Appalachians of southwestern Virginia, western North Carolina, and eastern Tennessee. Of the total spruce-fir type in the region, 74% occurs in the Great Smoky Mountains National Park, with the rest in the Black Mountains (11%), Balsam Mountains (10%), and on Roan Mountain (2%), Mt. Rogers/Whitetop Mountain (2%), and Grandfather Mountain (1%).

Southern pine and oak-pine forests in the southeastern United States occur on approximately 18 million ha in Virginia, North and South Carolina, Georgia, and Florida (Sheffield et al., 1985). These forests occur in three physiographic regions, with 60% in the coastal plain, and 40% in the piedmont and mountains.

4.2.1 Southern Spruce-Fir Forest Condition

In a review of historical disturbance histories of southern Appalachian spruce-fir forests, Wentworth et al. (1989) stated that the current age distribution and species composition of these forests are attributable to stand dynamics resulting from past disturbances. For example, past disturbances apparently favored regeneration of Fraser fir in some circumstances, and thus influenced the pattern of Fraser fir mortality caused by subsequent balsam woolly adelgid infestation. This implies that analysis of changes in growth, mortality, and species composition, plot and tree data should be stratified by factors associated with disturbance history.

Concern exists over red spruce and Fraser fir in the Southeast because of concentrated areas of standing dead trees and possible growth reductions of surviving trees. Dull et al. (1988) conducted an aerial and ground survey of spruce-fir forest types in the southern Appalachians. Four geographic regions were sampled (Balsam Mountains, 50 aerial and 30 ground survey plots; Great Smoky Mountains National Park, 82 and 30; Black Mountains, 68 and 33; Roan Mountain, 32 and 30). Photographs of randomly selected 0.8 ha plots were viewed in stereo to determine the total number of dead trees and the number of live spruce and fir trees visible on the plots. A random ground sample was also performed, and with the exception of a few plots in the Great Smoky Mountains National Park, each sampled ground plot corresponded to an aerial plot. Using this double sampling approach, regression analysis was used to estimate the number of dead and live by species for the aerial photo plots. Each ground point was established using a cluster of five BAF 37.5 sampling points. Dull et al. reported that 24% of the total area had more than 70% of dominant and codominant standing trees dead, 6% had 30% to 70% standing trees dead, and 70% of the area showed less than 30% standing trees dead (see Figure 17).

Since the early 1960s, southern Appalachian spruce-fir forests have shown increased Fraser fir mortality caused by an introduced insect, the balsam woolly adelgid. Following initial infestation, tree mortality occurs within nine years. The adelgid was first detected in the southern Appalachians on Mt. Mitchell, in 1957 (Speers, 1958). It subsequently has spread to Fraser fir on other mountains, although for unknown reasons mortality rates are lower on Mt. Rogers. The adelgid does not infest red spruce.

Red spruce occurs on approximately 44,500 ha of forest land in West Virginia. In a survey of red spruce forest types in West Virginia, Mielke et al. (1986) reported that only about 7% of the total land area of spruce type has greater than 10% standing dead. Approximately 35% of basal area, equivalent to 40% of the number of trees or 33% of the total volume, is dead or declining. Declining red spruce were defined as trees having live crown ratios of less than 40%. Most saplings and some pole-size trees were declining because of natural crown suppression. However, neither the actual nor the expected percentages of dead and declining trees that have less than 40% live crown ratios are known. The primary reason for decline of crowns for larger trees was branch dieback and flagging. *Valsa kunzei* was commonly associated with branch mortality.

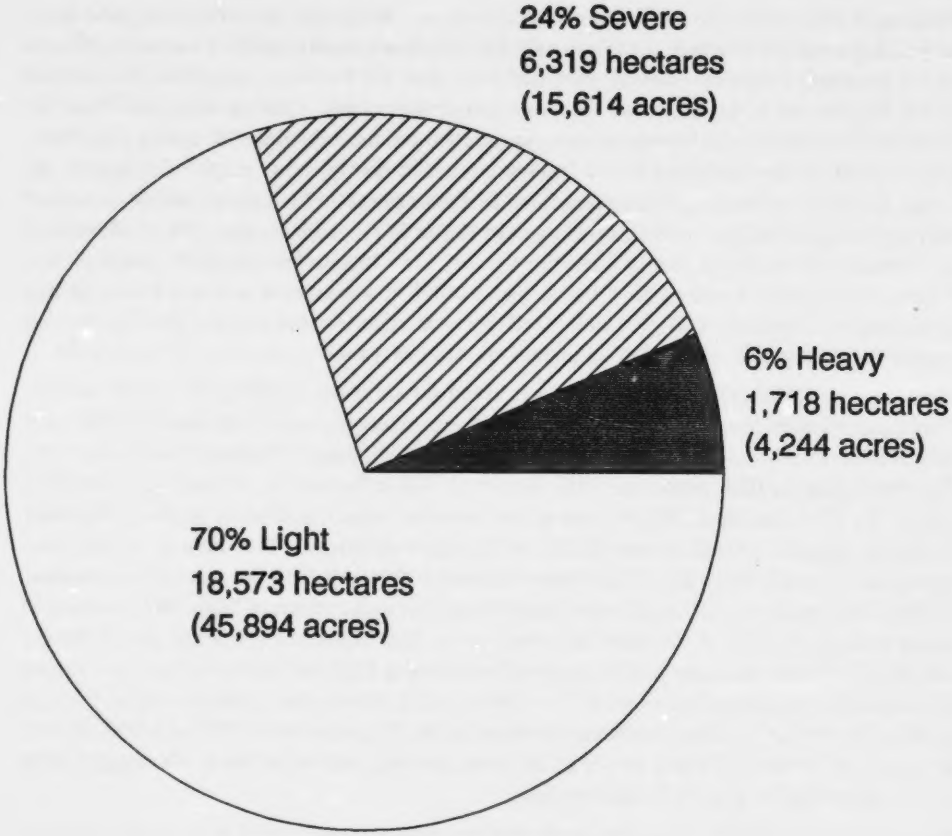

Figure 17. Spruce-fir forest type in the southern Appalachian Mountains classified by percent standing dead (redrawn from Dull et al., 1988).

V. kunzei is pathogenic and can be fatal to red spruce (Waterman, 1955). Although Mielke et al. (1986) did not conduct any radial growth trend analyses, *V. kunzei* may be contributing to the reduction in red spruce radial growth in West Virginia reported by Adams et al. (1985). Photographs taken during 1870-1920 (Clarkson, 1975) indicate that the conditions of some red spruce crowns appear similar to those of declining trees in Mielke et al.'s survey. This suggests that the current presence of *V. kunzei* is probably not a new phenomenon.

Mielke et al. (1986) also reported that red spruce seedling and sapling density increased with increasing levels of overstory mortality. Regeneration was visibly unaffected by overstory decline and mortality, and seedlings were apparently healthy.

Patterns of accumulated mortality (standing dead stems) throughout the southern Appalachians vary with species composition. Zedaker et al. (1989) analyzed stand conditions and site attributes on 142 randomly located permanent plots and more than 400 temporary sampling plots located on Mt. Rogers and in the Black and the Great Smoky Mountains. They reported that Fraser fir dominance increased with increasing elevation; spruce was most dominant at middle elevations, and a mixed spruce-hardwood forest (yellow birch, fire cherry, sugar maple) dominated the lowest elevations at which spruce occurs. Standing dead stems for both spruce and fir increased with increasing elevation. In 1987, standing dead Fraser fir ranged from over 90% of all stems at all elevations in the Great Smoky Mountains to less than 10% in low-elevation stands on Mt. Rogers. Red spruce standing dead was as high as 80% in some stands in which Fraser fir was dominant, but it typically averaged 10%. Zedaker et al. reported that standing dead stems may represent as many as 25 years of accumulated mortality in spruce but less than 15 years in fir.

Nicholas et al. (1989a,b) reported on changes in red spruce crown condition from 1985 to 1988. They sampled more than 2700 trees on 142 400-m^2 permanent plots at elevations from 1525 m to 1980 m in the Black and the Great Smoky Mountains and Mt. Rogers National Recreation Area. For all locations in 1985, more than 83% of trees at 1980 m had healthy crowns (i.e., over 90% intact). In 1988, less than 43% of trees at that elevation were classified as healthy. No other consistent changes over time were found at the other elevations. Tree crowns at mid- and low-elevation stands in the Black Mountains declined in 1986 and 1987, but recovered somewhat in 1988. Two ice storms occurred in the Black Mountains in the winter of 1986-1987, resulting in crown damage to 28% of the surviving overstory at high elevations (Nicholas and Zedaker, submitted). During the same period, at elevations between 1525 and 1830 m in the Great Smoky Mountains the number of trees with 90% to 100% of their crown intact decreased by an average of 20%. No decline in crown condition was found at Mt. Rogers between 1985 and 1988. Fraser fir mortality due to the balsam woolly adelgid was also reported to be low at Mt. Rogers even though the adelgid is present in high numbers.

The balsam woolly adelgid infestation in the southern Appalachians has left most mature fir dead. To maintain a continued presence of fir on the southern peaks, fir stems must be able to survive to seed bearing age and produce viable seed. Seed bearing of overstory fir usually begins at 20 to 30 years, with larger seed crops occurring at two- to four-year intervals. The majority of fir seed at all three southern intensive sites were empty in 1986 and 1987. However, Nicholas et al. (1989b) also found a range in germination capacity of filled seeds (both viable and rotten) averaged over the two collection years: Mt. Rogers, 31%; Black Mountains, 69%; and the Great Smoky Mountains, 86%. These values are well within normal ranges.

Fir seedling densities tended to follow overstory density patterns. Overstory fir increased with increasing elevation (Zedaker et al., 1989). Overall, seedling densities varied: the Black Mountains had 15,000 stems/ha; the Great Smoky Mountains had 16,300 stems/ha; and Mt. Rogers had 49,400 stems/ha. Continued long-term monitoring is necessary to identify the regeneration success of the southern spruce-fir forests; however, seedfall, viability, and seedling densities indicate an active regeneration process (Zedaker, personal communication).

The possibility of a red spruce decline has also prompted concern for the capability of spruce to produce viable seed. Overstory spruce usually begin seed bearing at 30 to 40 years of age, with larger seed crops occurring at intervals of three to eight years (USFS, 1974). As was found for fir, the majority of spruce seeds were empty for 1986 and 1987 at three southern intensive sites (Mt. Rogers National Recreation Area, VA; Black Mountains, NC; Great Smoky Mountains, NC and TN). However, the USFS (1974) reported an average germination capacity of 62% for red spruce filled seeds, and all three southern sites averaged higher germination capacity for filled seed: Great Smoky Mountains, 82%; Mt. Rogers area, 94%; and the Black Mountains, 96% (Nicholas et al., 1989b).

Spruce seedling densities also tended to follow overstory density patterns. Overstory spruce decreased with increasing elevation (Zedaker et al., 1989). Density of spruce seedlings (1 year old to 1.37 m tall) was lower in the Mt. Rogers (5,700 stems/ha) area than in either the Great Smoky Mountains (50,000 stems/ha) or the Black Mountains (51,700 stems/ha).

Silver et al. (1988) reported no visible red spruce crown decline symptoms from 1982 to 1987 at four of six southern sites (Whitetop Mountain, Monongahela National Forest, George Washington National Forest, and Shenandoah National Park). To study crown condition, a single 600-m transect was located in each stand. Transects were 25 m apart in elevation. The crown conditions of approximately 85 to 100 spruce trees were rated at each mountain. The 1982 transects were not permanently marked, so the 1987 transects are not identical, but they represented the same elevations. Crown condition may have changed at Roan Mountain and Mt. Mitchell from ice storm damage in the early winter of 1986 and winter of 1987, although statistical tests were not significant. Increased mortality was not detected at any of the six sites.

To investigate whether radial growth changes have occurred recently, the same tree-ring data from red spruce in the southern Appalachians were independently analyzed and interpreted by Cook (1988), Ord and Derr (1988), and Van Deusen (1988). The tree-ring data consisted of dated tree-ring sequences, two cores per tree, from several trees (commonly five or more) on 48 Tennessee Valley Authority (TVA) and 21 National Park Service (NPS) plots. Tree ring widths from the TVA plots were only available back to 1850. The NPS rings were dated back to 1900, but all NPS trees were at least 200 years old (C. Eagar, personal communication). An additional data set of two cores from each of 19 trees at Clingmans Dome, NC, was used by Van Deusen. Meteorologic data from 1931 to the present and stand and plot data from the 1985-1986 field seasons were also available.

Only 23 plots showed any evidence of the balsam woolly adelgid, and many of these plots had only slight damage. In the 48 TVA plots, a wide range of balsam woolly adelgid damage was found; half of the TVA plots had no Fraser fir trees. The 21 NPS sites had very little damage at the time of sampling, and no damage was found at the high-elevation NPS sites (E. Smith, C. Eagar, personal communication). Sites were not differentiated in analysis, however. Thus, the possible influence of the adelgid cannot be dismissed, because approximately one-third of the data contained Fraser fir mortality induced by the adelgid.

In each study, changes in the red spruce growth pattern over recent decades were found, although the nature of the changes differed with the analytical approach used to model ring width. Cook used a standard dendrochronological approach, applying negative exponential or linear detrending and multiple regression to quantify the effects of climate. Ord and Derr used Box-Jenkins time series models (Box and Jenkins, 1976) and spatial analysis. Van Deusen used the Kalman filter to quantify the time-varying influence of climate.

Cook found a steplike decline in radial increment since 1966, with the greatest decline occurring at the highest elevations (see Figure 18). At all elevations, apparent increases in red spruce sensitivity to drought since 1966 and to summer temperature since 1977 led Cook to suggest that southern red spruce are in a prolonged period of climatic stress. Similarly, Van Deusen concluded that since the 1960s, the trees have become more responsive to climate than they were previously (see Figure 19). Van Deusen suggested that the high mortality rate in associated Fraser fir caused by the balsam woolly adelgid, and the subsequent opening of the stands, might have been a catalyst for the increased sensitivity. Cook noted that the highest elevations (with the greatest growth reductions) contained the greatest proportion of Fraser fir and were most heavily impacted by the adelgid.

Ord and Derr analyzed TVA plots only, and they concluded that there was a reduction in average ring width in the 1960s, followed by stabilization at this lower level. Ord and Derr's analysis indicated that ring width decreased with increasing elevation. The most significant predictor of ring width was the proportion of live trees on the plot, which the authors interpreted to be an indicator of stand health. This variable may also be a surrogate measure of the balsam woolly adelgid. Ord and Derr's study was unique in that it incorporated a form of spatial analysis that accounted for geographic variables such as longitude, latitude, and elevation by correlating residual ring widths from a plot with residual ring widths from the nearest neighboring plots. Although Ord and Derr found evidence of a weak spatial association in these correlations, to a large extent the spatial correlation could be accounted for by the biotic variables considered (number of live trees, number of dead trees, stand basal area of live trees, stand basal area of dead trees, and derived indices based on these variables). Ord and Derr did not directly investigate the possible influence of climate.

In a separate dendrochronological study, McLaughlin et al. (1987) analyzed red spruce tree rings from 201 trees in 13 southern Appalachian forest stands. Eight of the stands were at Mt. LeConte, TN, two were at Clingmans Dome, TN, and the other three were at Mt. Sterling, NC, Whitetop Mountain, VA, and Monongahela, WV. Two cores were extracted from opposite sides of each tree, for usually 15 trees per stand, and then crossdated. The ring-width sequences for each tree were standardized to remove low-frequency growth shifts that were not well replicated across a stand. A negative exponential curve was used to standardize, and, if this curve did not fit, simple linear regression was applied. Mean chronologies for each stand and across stands were produced, providing a record of approximately 200 years of spruce growth.

McLaughlin et al. then estimated the relationship between radial growth and climate by correlating ring width and the climatic variables of monthly mean temperature and total precipitation for 16 months beginning June of the year prior to, and ending with September of the year of, ring formation. Precipitation did not consistently improve model performance. The most frequently included temperatures were from late summer of the year prior to ring formation, with negative coefficients; next were those of winter (October-January) with positive coefficients, and February, with a negative coefficient. The latter was interpreted as the potential for early spring thaw-freeze cycles to damage buds and young foliage.

Data from 1900-1940 were used to estimate the climate-growth model, and data from 1941-1960 were used to verify the model. Only 5 of the 13 sites verified the model for 1941-1960. The fitted regression was then applied to post-1960 data to investigate whether growth trends for that period could be predicted with the past climate-growth relationship; only two sites verified the model for 1961-1980. Mean differences indicated that the models typically underestimated ring widths for 1941-1960 and overestimated growth after 1961. McLaughlin et al. concluded that red spruce is growing more slowly than past climate-growth relationships would predict, and that this

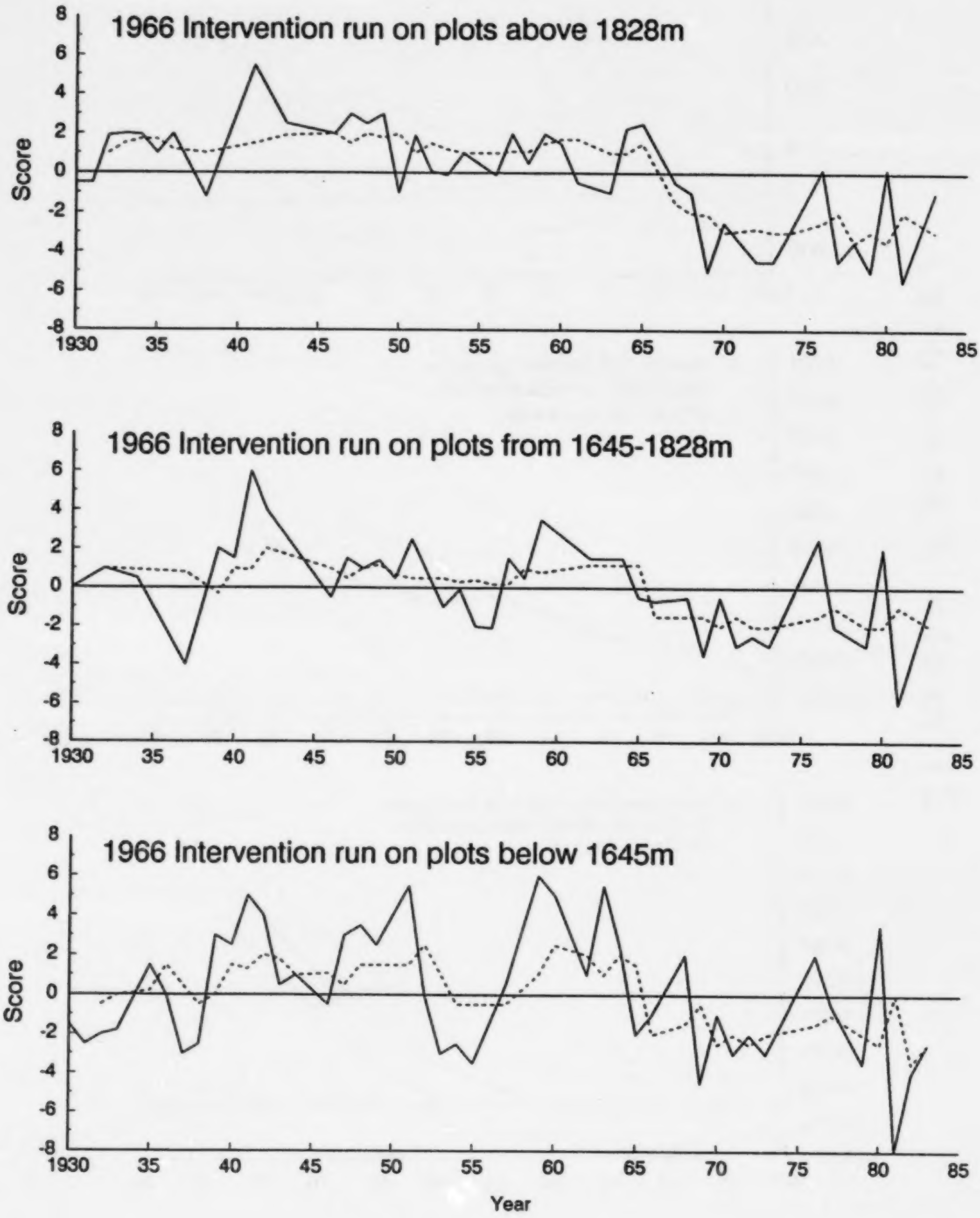

Figure 18. Actual (solid) and estimated (dashed) tree ring scores derived by fitting a step intervention model to each series. The date of the intervention is 1966 (redrawn from Cook, 1988).

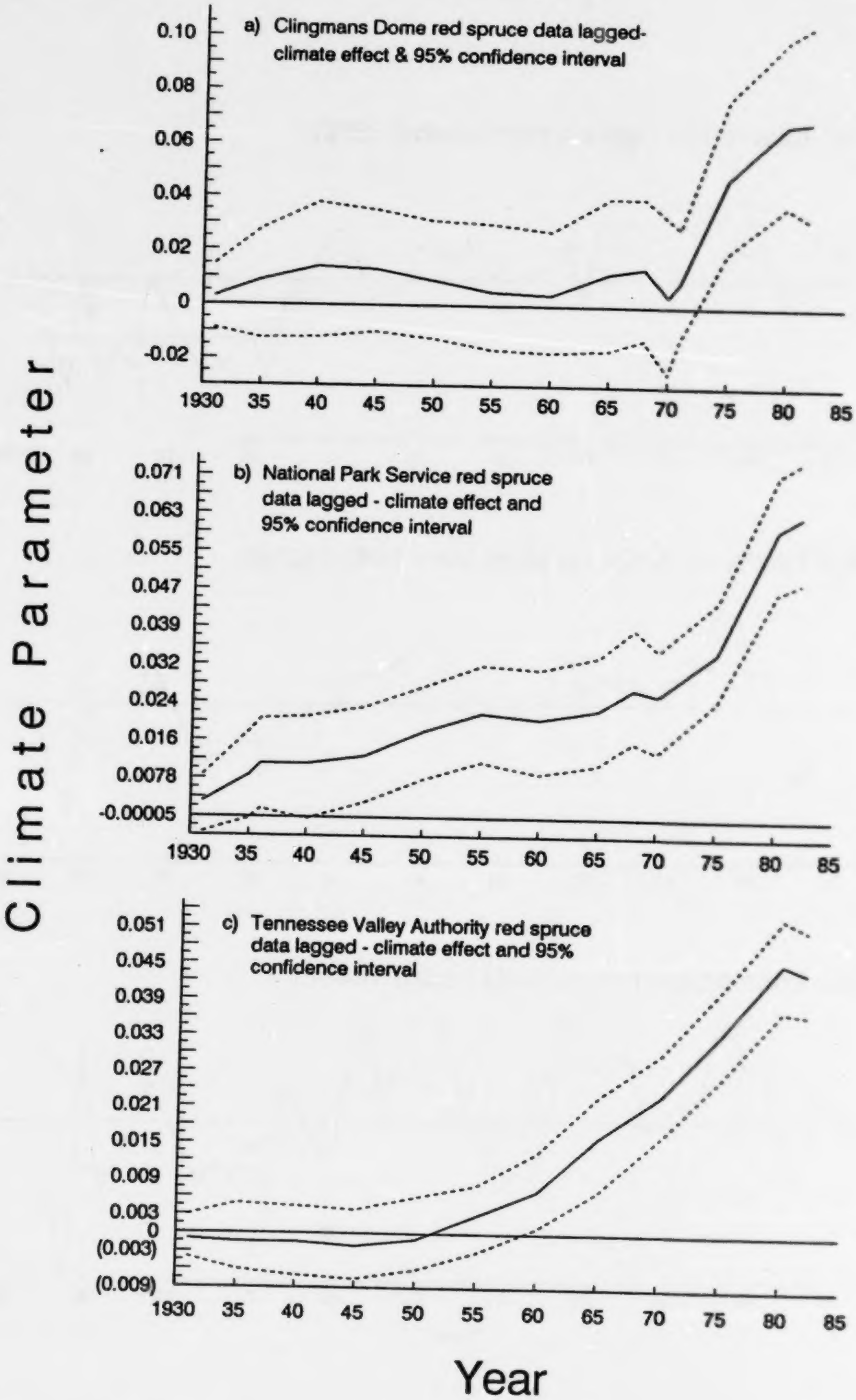

Figure 19. Time trend predictions of the climate variable, which is a linear combination of monthly average rainfall and temperature variables for a) Clingmans Dome, NC; b) National Park Service; and c) TVA sites (redrawn from Van Deusen, 1988).

indicates that red spruce is responding to factors other than climate or to climate in an unusual way. The reduction in growth was more pronounced at high elevations.

In their report, McLaughlin et al. (1987) suggested that growth of their sampled trees should be independent of age because diameter growth of dominant trees in virgin stands has been shown to be largely independent of age (USDA, 1917). However, a review of the mean age and standard errors reported by McLaughlin et al. suggests that tree ages of dominant and codominant trees at a site vary considerably, suggesting that some stands were relatively young (e.g., Monongahela, WV; average age = 84, standard error = 20), while others were comprised of both older and younger trees (e.g., Mt. LeConte, TN; average age = 253, standard error = 83). Comparisons of McLaughlin et al.'s trees with those of virgin old growth trees seems questionable because they indicate that a number of sites showed evidence of competitive release over the past 100 years at a few lower elevations. The large standard errors for age (McLaughlin et al., 1987) at some high-elevation sites plus recent work by Van Deusen (in press) for high elevations suggest possible releases at some high-elevation sites as well.

With respect to insects and pathogens, the balsam woolly adelgid has had a strong impact on southern Appalachian spruce-fir forests. In the past decade, more than 40% of the canopy in the high elevations has died as a result of adelgid infestation (Nicholas et al., 1989a,b). More than 45% of all remaining live canopy fir were infested by 1988. Red spruce in fir-dominated areas may be susceptible to shock of sudden exposure, similar to thinning shock (Zedaker et al., 1989). However, thinning can be beneficial because there is less competition for light, water, and nutrients. Tree and stand response to thinning depends on the current structure of the stand and the degree of thinning.

Bruck (submitted) assessed tree condition and the presence of insects and pathogen infestation, particularly of roots, during 1985-1987 in the southern Appalachians. A remarkable result was the apparent lack of insect infestation and absence of severe pathogens in the study areas. However, the balsam woolly adelgid was frequently found and, as discussed previously, the adelgid is the principal cause of Fraser fir mortality (Groton and Eagar, 1988; Dull et al., 1988).

Hartman et al. (submitted) found that densities of the parasitic nematodes *Sphaeronema sasseri* and *Pratylenchus macrostylus* in soil-root samples were positively correlated with elevation in the southern Appalachians ($r = 0.49$, $p < 0.01$). They also observed that tree decline indices (assessed via crown defoliation) were positively correlated with elevation ($r = 0.69$, $p < 0.01$ for southern Appalachian Mountains; $r = 0.88$, $p < 0.01$ for the Black Mountains). When nematode densities were compared with tree decline indices, however, Hartman et al. observed weaker positive correlations between nematode densities and decline across all studied areas ($r = 0.22$, $p < 0.02$), and for high elevations in the Black Mountains, a negative correlation was reported ($r = -0.79$, $p < 0.01$). In spite of the instance of negative correlations, the authors felt the role of nematodes in the etiology of forest decline warranted further investigations.

In a study of soils in southern red spruce forests, McLaughlin et al. (in press) reported higher P, K, and Al levels, and lower Ca levels in soils at 1935 m compared with soils at 1710 m at Clingmans Dome, NC. Reduced soil Ca:Al ratios at the higher site were reflected in lower Ca:Al ratios in the foliage. At the higher site there was evidence of radial growth reduction, and foliar concentrations of P, Mg, Ca, and K were lower. This study thus indicates an association between reduced forest vigor and decreased soil nutrient availability.

Summary

Since the mid-1960s, radial increment of red spruce at the highest elevations has reduced slightly. Climate, pollutant deposition, and the balsam woolly adelgid have been suggested as contributors. The adelgid is most severe at high elevations because of increased Fraser fir densities at higher elevations, and thus the possible influence of the adelgid on radial growth of red spruce should not be overlooked. The possible effects of past disturbances and resulting influences of stand dynamics on red spruce radial growth have not been thoroughly investigated. Radial growth and pollutant exposure could not be correlated in these studies because deposition estimates and tree-ring data were obtained from different sites. In addition, many tree-ring analyses were conducted prior to the initiation of deposition monitoring. However, it should be noted that many pollutants are present for longer durations and at higher concentrations as elevation increases.

In the southern Appalachians, Fraser fir mortality caused by the balsam woolly adelgid has increased since the early 1960s. In spruce-fir forests of this region, 24% of the total area has more than 70% of dominant and codominant standing dead trees, and 70% of the total area has less than 30% standing dead trees. Areas with the greatest standing dead were dominated by Fraser fir that had been killed by the adelgid. Red spruce standing dead typically averaged less than 10% throughout the region, and recent mortality events have been associated with ice storms. In West Virginia, only 7% of the total land area of spruce type has greater than 10% mortality.

Red spruce crown condition has generally declined from 1985 to 1988. Two ice storms in the winter of 1986-1987 resulted in crown damage to 28% of surviving trees at the highest elevations. During the same period at lower elevations, the number of trees with more than 90% of their crown intact decreased by 20%. In a survey in West Virginia, 40% of the red spruce trees at elevations above 1000 m had less than a 40% live-crown ratio, with most of the low crown ratios associated with natural crown suppression.

4.2.2 Southern Pine Forest Condition

Using the FIA data described in Section 3.4, Sheffield et al. (1985) provided the AARI by two-inch diameter classes, along with sample size and standard error, for growth that occurred between the third and fourth and between the fourth and fifth FIA inventories. Planted and natural stands were distinguished. The AARIs were compared by diameter class for each combination of species, stand type, state, and region (Piedmont, Coastal Plain, and Mountain). For Piedmont and Mountain regions, Sheffield et al. observed reductions in diameter growth for all but the largest pine trees (see Figure 20). The reduction varied across species and diameter class, but it averaged 20% to 30%. The majority of differences in the AARIs were significant using an α level of 0.32 (i.e., one standard error for a two-sided test). Trees in the Coastal Plain did not show such a clear pattern of growth reductions.

Believing that differences of the magnitude seen in the Piedmont and Mountain regions were unusual, Sheffield et al. investigated the history of southern pine AARI trends. They concluded that AARIs of southern pine species in natural stands throughout the regions had decreased by 30% to 50% between the third (1957-1966) and fifth (1977-1985) surveys. Trends in planted stands were inconsistent, showing increases, decreases, and stable increments within the regions.

The authors did not make causal inferences, but they did suggest possible explanations needing follow-up investigations, including: 1) changing stand structure (age, stand density, site quality, and hardwood competition); 2) climate, especially drought; 3) fewer old-field sites (i.e., fewer cropland sites available for conversion to forests); 4) increased incidence of disease; and 5) atmospheric deposition.

In discussing these follow-up investigations of the southern pine decline, four issues will be considered:

1) Is there a general radial growth decline or are there localized declines?
2) What independent variables most affect radial growth increment, and how well have they been measured or estimated?

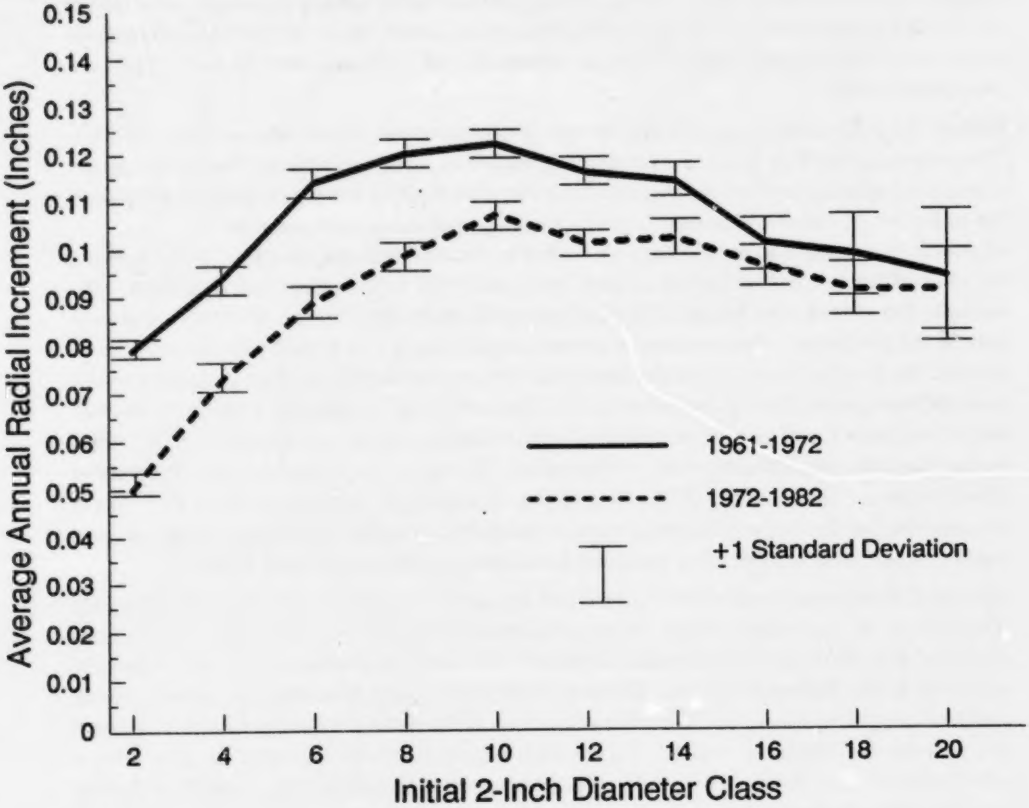

Figure 20. Average annual radial increment by diameter class for loblolly pine growing in natural stands in the Georgia Piedmont and Mountain regions (redrawn from Sheffield et al., 1985).

3) What is the influence of these independent variables?

4) Are radial growth declines of individual trees carried over into per unit area productivity?

To answer these questions, the data and analyses must meet certain criteria. First, important stand covariates associated with radial or per unit area growth must be measured. For southern pines, these covariates include age, site index, total stand density, hardwood density, and perhaps crown ratio. Second, we must either track the same trees and plots over time or identify plots that in the next time period would be comparable to the current sample of trees and plots (i.e., plots having a similar range of covariate values for the two time periods). Third, we need information about climatic conditions that may precondition tree response, such as site-specific rainfall and temperature patterns over the growing season and cumulative information about climatic conditions over several seasons. Finally, spatial or geographic information and analyses are needed so that local versus regional differences in growth can be identified. Information about elevation, longitude, and latitude are important, and analyses should be performed on a plot-by-plot basis.

Most of the data available for studying changes in southern pine growth rates are from the FIA. The purpose of the FIA is to estimate standing volume of trees, and although the design allows changes in populations of trees and stands to be detected, the FIA was not designed to determine the causes of any observed changes in per unit area growth rates. Investigations into the causes of growth rate changes using FIA data are limited by the fact that important tree, stand, and site variables either are not measured or have been measured in different ways over time. For example, the second criterion listed above is not met in analyses of FIA-based basal area growth data in the Southeast: Comparisons of growth rates between two growth periods are limited because the samples are not from the same plots and are nearly independent, random samples from different populations at two points in time (Bechtold et al., submitted). Changes in average age, stand density, and amount of hardwood competition in natural pine stands have occurred during the time periods compared, and therefore the explanatory variables are confounded (Sheffield et al., 1985; Hafley, 1985). Quantifying changes in forest structure from FIA data is appropriate, but the large variation in characteristics that describe this structure (e.g., age and density) contributes to large error variances for estimates of change (Hafley, 1985).

In initial FIA analyses (Sheffield et al., 1985) and subsequent supporting analyses of the FIA data (Bechtold et al., submitted; Ruark et al., submitted; Zahner et al., 1989), the authors have suggested that there have been general declines in both radial and basal area growth, apparent especially in the Piedmont region. However, local versus regional changes in growth (either positive or negative) cannot be differentiated because all the data have been pooled into regional averages for the Piedmont region. Future work should investigate changes in growth on a plot-by-plot basis to determine if groups of plots in a local area are showing changes, or if plots over the entire Piedmont region are changing.

Decades of growth and yield research in the South have shown that stand structure variables such as age, density, site quality, and hardwood competition are clearly important in predicting growth and yield (MacKinney et al., 1937; MacKinney and Chaiken, 1939; Coile and Schumacher, 1964), as is climatic variation (Zahner et al., 1989). As Hafley (1985) has noted, the FIA data should be used cautiously when interpreting statistical significance or when making causal inferences for these variables. The reasons for this are not immediately apparent, and we will discuss them briefly. The FIA data are used to estimate average age, stand density, site quality, hardwood competition, and rates of growth. However, methods of measuring these variables have been applied differently over time, and therefore interpreting the temporal changes in these variables is difficult. Quantifying the effect of these variables on growth rates is an even more difficult if not intractable problem.

Zahner et al. (1989) used FIA-based data to evaluate the influence of these independent variables on radial growth increment. The data consisted of two increment cores from opposite sides of the bole for at least five loblolly pine trees in 131 even-aged, natural, well-stocked stands in the Piedmont region of Georgia, South Carolina, and North Carolina. The cores were measured within 0.01 mm and cross-dated. Selection criteria allowed only plots from undisturbed stands of loblolly pine. The plots were from 25 to 80 years old in 1984, with stand density in the normal range of basal areas and stocking levels for typical pine sites of site index between 18 and 28 m (base age 50). Each plot had been inventoried in three previous FIA surveys and was last measured in 1985. Hardwoods were present as understory and midstory in all inventoried stands, and as a component of overstory in some stands.

Zahner et al. (1989) applied multiple regression to these data to estimate a variant of the linear aggregate model (Graybill, 1982) and thereby investigate the influence of age, stand density index, climate, site index, and calendar year on ring width. All interaction terms were nonsignificant, and the final regression equation included the following predictors: stand density index, climate, year, and the reciprocal of age. Climate was defined as the three-year moving average of Drought Index for Southern Pines values (DISP)(Zahner and Grier, 1989) averaged over two previous years and the current year. (DISP values greater than 1.0 are associated with wetter than normal years, and values less than 1.0 with drought conditions.) Ring width declined with age, stand density, and year, and it increased with DISP values.

Zahner et al.'s results indicate a 38% decline in radial growth from 1949 to 1984 for the 131 plots located throughout the Piedmont. The radial growth reduction appeared not to be associated with changes in either site productivity or stand ages. The authors suggested that Piedmont upland sites are susceptible to drought as an ongoing predisposing stress (Manion, 1981; Zahner and Myers, 1986), and that exceptionally dry years in the mid-1950s may have started a regional decline in radial growth.

The radial growth decline was partially associated with changes in stand density within age classes. Zahner et al. (1989) reported that stand basal area increased over time. This result indicates either that conditions in later years permitted or favored more stems per hectare or that conditions in earlier years discouraged higher density. Possible factors could include changes in thinning practices or density control (e.g., hardwood competition). Tree-ring simulations indicated that this change in average stand density with time would be expected to result in about a 12% decrease in ring widths, which leaves a 26% decline in radial increment that could be associated with the passage of time (see Figure 21). Zahner et al. concluded that the reduction in radial growth exceeded what could be attributed to climate and stand structure changes, and they suggested that time-related exogenous factors such as O_3 or other pollutants were operating at the regional level.

Another possible explanation for the decrease in radial growth not accounted for by climate and stand structure is that the variation from increased stand density within age classes over time was not accounted for in the regression model by a simple linear combination of stand density index and the passage of time. Zahner et al. (1989) reported that the interaction of stand density index and calendar year was highly significant. When used in lieu of stand density index and calendar year separately, the interaction did not provide a better fit than the two independent effects in the linear model. However, in testing the interaction terms, it is not clear that substituting variables resulting in no change in R^2 would necessarily also mean no change in percent decreases in radial width. Thus, a lack of change in R^2 is perhaps not the best criteria for formulating the eventual prediction model.

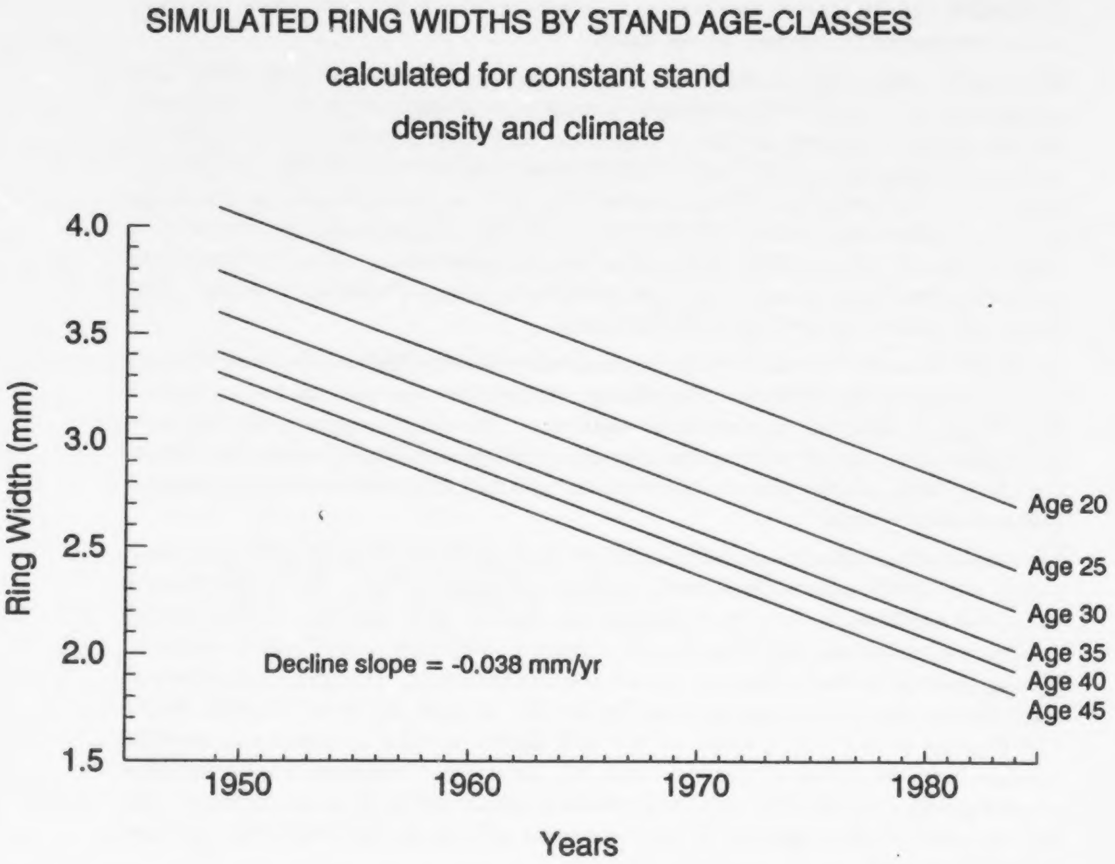

Figure 21. Simulated loblolly pine ring widths by age class over time from 1949-1984 (redrawn from Zahner et al., 1989).

Data for evaluating southern pine growth rates only exist for the last three to four decades. Because a baseline rate of growth has not been established, abnormally high prior growth rates cannot be distinguished from abnormally low recent growth rates. Research at Duke Forest by Christensen and Peet (1988) provides evidence for a cyclical nature of loblolly pine growth rates. They studied 30 permanent plots that had been established in 1933 and 1934. Tree growth increment (new volume or biomass/tree/yr) was negatively correlated with stand density ($R^2 = 0.88$) such that as stand density declined, growth increment increased. Christensen and Peet then used residuals from a regression model of this relationship to look for departures from the expected growth trend over time. They found significant temporal trends in the residuals that suggested the following cyclic pattern of growth rates: After adjusting for stand density, growth increment during 1965-1984 was less than from 1934-1965. However, since 1984, growth rates have increased. Comparisons between climate data and these growth patterns suggest no obvious correlation.

Christensen and Peet also reported that the temporal signal was present only for plots that were allowed to thin naturally. Artificial thinning greatly altered the growth-density relationship. They suggested that silvicultural treatments may greatly increase the variance in growth increment observed in extensive data sets such as the FIA data, and that any systematic and widespread changes in silvicultural practices over time could result in systematic trends in growth increment.

Warren (in press) used a simple conceptual model to suggest that most of the radial growth reductions of loblolly pine reported by Sheffield et al. (1985) can be explained by hardwood competition. Hardwood competition can be expected to reduce pine growth (MacKinney et al., 1937), and hardwood component has increased in southern pine stands over the last two to three decades (Hafley, 1985; Sheffield et al., 1985). Warren concluded that the application of his conceptual model to data summaries in Sheffield et al. (1985) supported the possibility of growth rate reductions in the smaller diameter classes of natural southern pine stands, with more severe reductions in the Piedmont and Mountain regions than in the Coastal Plain, and he suggested that most of this reduction could be explained by hardwood competition. Warren also stated that it is reasonable to expect that smaller pine trees are under greater hardwood competition than older and larger pines, which appears to describe the trend observed by Sheffield et al. (1985) of a smaller radial growth decrease for larger trees than for smaller trees.

As indicated previously, the influences of important tree and stand variables such as hardwood competition are difficult to quantify and assess from the FIA data given that samples were not from the same plots and were nearly independent, random samples from different populations at two points in time (Bechtold et al., submitted). Average age, stand density, and amount of hardwood competition in natural pine stands are known to have increased during the time periods compared. When levels of covariates are not comparable between time periods, adjustments of covariate levels to a common value are unlikely to be meaningful or interpretable.

Finally, are radial growth declines of individual trees carried over into per unit area productivity? Bechtold et al. (submitted) used Southeast FIA data to compare stand-level basal area growth between the two periods from 1961-1972 and 1972-1982. Regression models were fit to adjust for stand age, stand density, mortality, hardwood competition, and site quality. Annual basal area growth rate was lower in the most recent period than in the earlier period, even after adjusting for temporal differences in the predictor variables. Bechtold et al. concluded that changes in stand structure were not responsible for the recent changes in growth rates. A similar study of loblolly and shortleaf pine in Georgia and Alabama produced similar conclusions (Ruark et al., submitted). However, interpretations of the FIA-based growth analyses are questionable because independent surveys of different populations over time result in uncertainty in the comparability of the relationship of age and density with growth rates over time.

In a study of southern pines that did not use FIA data, Lloyd and Waldrop (in preparation) compared stand-level periodic growth while adjusting for the effects of age, site, and stand density. Their goal was to look for growth trends over time that were not explained by stand dynamics. Lloyd and Waldrop investigated basal area increment from natural loblolly pine stands on highly productive sites in coastal Virginia and South Carolina. The data were from 72 permanent 1/10-ha fixed-radius growth and yield plots periodically remeasured over 28 years. Average site index (base age 50) was 30.5 m (with a range from 21 to 37 m), average initial stand age was 38 years (range 21 to 55 years), and average initial basal area was 9 m^2 (range 4 to 15.5 m^2). The rules for thinning were not documented in the study plan or establishment reports, but thinnings ranged from light to heavy, with the goal of producing a range of stocking over a range of ages.

Growth was defined as average annual basal area increment of survivor trees from five-year remeasurement periods. The earliest and most recent remeasurement periods (periods 1 and 6) were compared; period 1 included the 1955-1959 growing seasons for all plots, and period 6 covered growing seasons 1976-1980 for Virginia and 1977-1981 for South Carolina. There was no mortality during period 1, and only a few dead trees in period 6, which indicates that thinning generally maintained sufficient growing space. Growth values from two plots in coastal South Carolina were unavailable for period 6 because they were destroyed by bark beetles prior to 1977; thus, there were 72 observations for period 1 and 70 observations for period 6.

Lloyd and Waldrop used two analytical approaches. First, they compared mean basal area growth in periods 1 and 6 for plots of similar age. Only plots with common age ranges for the two periods were used. As a result, mean annual basal area growth was compared for 29 period-1 and 27 period-2 plots that were 43 to 55 years old. Average ages were nearly identical (the earlier period averaged one year older). The stands in period 6 were on slightly more productive sites, with average site index 1.8 m greater. Periods 1 and 6 had similar mean initial basal areas (25.5 versus 24.8 m^2/ha).

A t-test indicated a significant period-6 growth reduction of 0.09 m^2/ha/yr ($p > 0.01$). A covariance model using site index estimated a period-6 reduction of 0.11 m^2/ha/yr (standard error = 0.1323). Initial basal area was included in the covariance model but was not statistically significant. Further statistical analyses verified the absence of a growth-density relationship. Lloyd and Waldrop suggested that the lack of a growth-density relationship may have resulted from the narrow range in stand age, and perhaps from disruption of the relationship due to thinning. Most stands were thinned more than once, with less thinning in more recent periods. The lack of significance for stand density could indicate that thinnings successfully allocated space for growth throughout the study. This seems likely because of the wide ranges in stocking and the limited mortality in both time periods. There appears to be no obvious confounding of thinning effects that would favor growth in the early period or reduced growth in the most recent, although, as just noted, thinning was not as frequent in the most recent periods. The effects of thinning are still being investigated.

Lloyd and Waldrop also compared growth in all study plots during periods 1 and 6 with growth predicted by a basal area growth model for managed natural stands of loblolly pine (Nelson, 1964). This model had been developed using period-1 data from the plots described above plus an additional set of plots in the Georgia Piedmont. The period-1 information used in this analysis comprised part of Nelson's data, so, as expected, observed growth was predicted with a high degree of accuracy. Comparing observed growth in period 6 with predicted growth from the model indicated that observed growth in the most recent period was far below expected.

However, with the addition of the Georgia Piedmont data, it is not clear how similar the ranges for each covariate were for periods 1 and 6, so the legitimacy of the comparison is unknown.

Although Lloyd and Waldrop did not directly analyze climate, general comparisons of DISP values indicated that climate was similar for the two periods. For South Carolina data, both time periods fell in the moderately less than average class (DISP = 0.899 for 1955-1959, and 0.876 for 1977-1981). The Virginia data were in the average growth class, with DISPs of 0.952 for period 1 and 1.085 for period 6. A possible drought explanation is even more unlikely given that period 1 had a severe two-year regional drought from 1954-1955.

In looking for potential predisposing weather events, Lloyd and Waldrop found that a large-scale regional ice storm occurred just after the first remeasurement. Crowns of trees were damaged, and recovery may never occur. Additional analyses are being performed to look for long-term effects (Lloyd, personal communication). The lack of epicormic buds in southern pines means that foliar losses due to ice damage that breaks off limbs or tops can be replaced only at the meristem (Hafley, personal communication). Depending on the severity of the damage, photosynthate production and subsequent tree vigor are reduced. At a stand level, the potential to respond to a release due to mortality of individual trees is never realized. In addition, there is normally a 3% reduction per year in loblolly pine basal area increment once the maximum growth rate has been attained, at approximately 10 years of age (Hafley, personal communication).

Smith (1989) compared annual survivor growth of plots distributed throughout the natural range of loblolly pine using data from the Southwide Pine Seed Source provenance study (Wells and Wakeley, 1966). A total of 15 plantations were planted in 1952 and 1953 in 10 counties distributed across Texas, Arkansas, Alabama, Georgia, South Carolina, and 1 parish in Louisiana. Each plantation was a randomized complete block design with four blocks. Although 15 seed sources were used in plantation establishment, only local seed sources were included in the analysis for two reasons: 1) to maximize the relevance and interpretability of results to the natural forest; and 2) so that off-site declines of some seed sources would not be confused with declines that may be associated with increased pollutant levels.

The purposes of Smith's analyses were to detect differences in growth rates by location and to indirectly evaluate atmospheric deposition's influence relative to other factors that may alter growth. Because atmospheric deposition measures were not available, Smith hypothesized three scenarios from which he could infer a pollutant effect: 1) no pollutants at any location; 2) equal pollutant levels at all locations; and 3) unequal levels of pollutants at some or all locations. Scenario 1 served as the null hypothesis and scenario 3 was the most likely alternative.

To determine the most plausible scenario, Smith correlated annual survivor basal area growth to plot-level variables (age, initial basal area/acre, site index, and mortality). Site index became nonsignificant when mortality was added, and it was therefore dropped from the model. Interactions did not improve the model. Because of the wide range of locations, climate variables were introduced. The best climate predictor was the annual minimum value for the Palmer Drought Severity Index (PDSI)(Palmer, 1965). The year introducing the greatest drought stress had the most influence on growth.

To determine if the models were consistent across location, thereby implying no pollutant effect, Smith compared the slopes at each location to a grand slope (i.e., a slope across all sites) for all covariates. The growth intervals started at age 10 (1963) and were based on four remeasurements, with the last in 1982 at age 30. A total of 148 growth observations from six locations were used for model estimation. When the slope for the location differed from the grand slope, an interaction was indicated; three significant interactions at three separate locations were identified. The interaction terms were associated with greater than average growth at two locations and less than average at a third (Clark County, AR). For this latter site, basal area mortality was the covariate, and it was associated with a relatively constant decline. Smith indicated that this decline was as likely to be associated with disease as with pollutants. He concluded that the number of interactions, their direction, and the variables involved were neither consistent nor of a magnitude greater than would be expected from natural variability. Although a wide range of locations were included, the cause of the three significant interaction terms is still unclear because there were relatively few observations. However, one possible conclusion is that the relationship between survivor basal area growth and stand dynamics (age, initial basal area, climate and recent mortality) is consistent across locations. This may indicate that the expected different levels of pollutants at the study locations are not influencing survivor growth.

Summary

In summary, comparisons of the last three surveys of natural southern pine species on nonindustrial private land indicate a reduction in average radial increment in the period between 1957-1966 and 1977-1985. Although most of the growth reduction has occurred on nonindustrial private forest land, some reductions have also been identified on public and industry land (Sheffield et al., 1985). Attempts to correlate changes in stand structure with individual tree and stand growth have not been fully successful because of limitations in data sets. Stand structure effects are not comparable between the periods of interest. There is some evidence that loblolly pine growth rates may be cyclic. Research on growth and yield plots at the Duke Forests indicate that after adjusting for stand density, data from permanent plots that were established in the early 1930s showed less growth increment during 1965-1984 than from 1934-1965, and, since 1984, growth rates have increased (Christensen and Peet, 1988). Comparisons with climate data suggested no obvious correlation. A study that used non-FIA data suggests that loblolly pine basal area increment and stand level covariates were generally consistent across study sites. Thus, atmospheric deposition may not have a detectable influence on survivor growth or on growth-stand dynamics relationships.

Individual tree growth reductions have not been firmly established, nor have any studies clearly identified a decrease in productivity per unit area. The radial growth reduction of diameter classes of trees (Sheffield et al., 1985) is expected given that the population of southern pine trees has changed over the last three decades. The trees from early decades were from lower density younger stands, but the most recently sampled populations represent higher density older stands. The populations of trees differ substantially between the time periods studied with respect to the important concomitant variables (i.e., the data ranges do not or only slightly overlap). Thus, covariance models should be used with caution because adjusted means may require substantial extrapolation of the regression lines to a region where only a few data points exist (Neter and Wasserman, 1974). The individual tree radial growth reductions identified by Zahner et al. (1989) were based on data similar to the FIA data, and radial growth reductions similar to those reported by Sheffield et al. (1985) were apparent in the data as well as in the analysis.

4.2.3 Wet Deposition

Spatial variation in the 1985-1987 precipitation deposition of H^+, SO_4^{2-}, NO_3^-, and NH_4^+ in southeastern states is presented in Figure 22. Sites in Tennessee, Virginia, and Kentucky had higher deposition of H^+ and SO_4^{2-} (300 to 600 eq/ha/yr) than the southernmost sites in Mississippi, Alabama, Florida, and southern Georgia (200 to 300 eq/ha/yr). North Carolina sites received an intermediate H^+ and SO_4^{2-} wet deposition of 300 to 400 eq/ha/yr. The precipitation deposition of NO_3^- and NH_4^+ was about 30 to 200 eq/ha/yr at most of the southeastern sites. Precipitation pH was generally between 4.3 and 4.7 in the Southeast, with SO_4^{2-} concentrations between 1 and 2.5 mg/L.

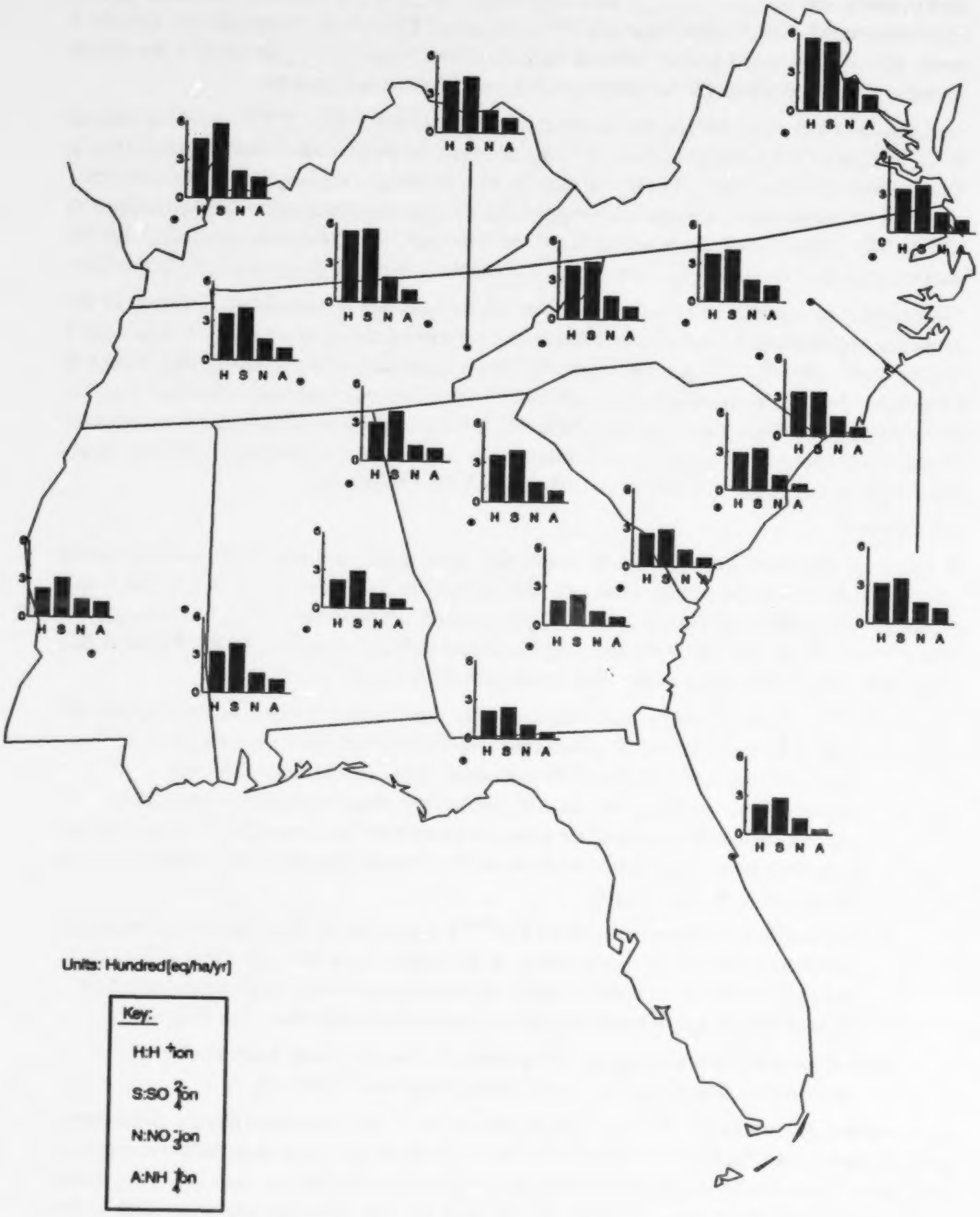
6
3
0
H S N A
Units: Hundred[eq/ha/yr]
Key:
H:H $^+$ion
S:SO $_4^{2-}$ion
N:NO $_3^-$ion
A:NH $_4^+$ion

Figure 22. Atmospheric wet deposition via precipitation for southeastern NADP sites, 1985-1987.

The variability in wet deposition between nearby sites (e.g., in eastern Tennessee and western North Carolina) appeared smaller than the differences between sites farther apart. However, the sites in Tennessee, Virginia, and North Carolina were generally at higher elevations than sites farther south and east, so spatial patterns may reflect elevational differences. Mohnen (1988a) reported that SO_4^{2-} deposition via precipitation exceeded 600 eq/ha/yr for two sites at elevations above 600 m. Increased precipitation at high elevations would be consistent with the spatial patterns in wet deposition for the Southeast that are indicated in Figure 22.

An analysis of wet deposition in southeastern forests suggested values of 300 to 400 eq/ha/yr of SO_4^{2-} and 150 to 200 eq/ha/yr of NO_3^-, with higher values in Virginia and North Carolina than in Texas (Shadwick et al., 1989). The lowest average monthly pH and the largest deposition typically occurred during summer. Limited data suggest that dry deposition accounted for approximately 25% to 50% of the total S and N deposition in the Southeast. Shadwick et al. concluded that the data were too uncertain for definite conclusions about the relative levels of wet and dry deposition.

With respect to elevational variation in acidity, cloud base was frequently near 1000 m in the southern Appalachian Mountains, and there were substantial chemical inputs from cloud water interception. The MCCP has monitored cloud water chemistry at Mt. Mitchell, NC, Whitetop Mountain, VA, and Shenandoah, VA, and the IFS has collected cloud water chemical samples in the Great Smoky Mountains National Park, NC. Because significant differences in cloud water deposition between the Southeast and Northeast have not been documented, cloud water inputs to southeastern regions were discussed with results for the Northeast.

4.2.4 Ozone

A subset of sites have been chosen to emphasize north-south and east-west gradients in O_3 concentration across the southcentral and southeastern growth trend areas (see Table 3 and Figure 2): Dickey Ridge in Shenandoah National Park (NP), VA; Hanover, VA; Lewisburg, WV; Wake Forest, NC; Davie, NC; Johnston, NC; Cumberland, NC; Greenville, SC; Mt. Pleasant, SC; Muscogee, GA; and Rockdale, GA. The results in Tables 5 and 6 indicate that:

1) O_3^{med} concentrations at high-elevation sites such as Dickey Ridge in Shenandoah NP ranged between 40 and 60 ppb during May through October. On average, O_3^{med} exposures appeared similar at the remaining sites (median diurnal hourly concentrations ranging from 10 to 65 ppb). Sites along the southern coast and towards the southern end of the study area may have had lower O_3^{med} concentrations than elsewhere (e.g., diurnal medians at Mt. Pleasant ranged from 5 to 48 ppb, and at Muscogee from 5 to 55 ppb).

2) daily maxima in O_3 concentrations (O_3^{worst}) at remote, rural, or agricultural sites did not exceed 120 ppb regularly, except at Rockdale where concentrations as high as 165 ppb have been recorded in June. Rockdale and Dickey Ridge experienced the highest percentage of hours with hourly concentrations greater than 80 ppb.

3) most sites had O_3 scavenging during the nighttime, even sites designated rural/agricultural. The exceptions were Dickey Ridge and Lewisburg.

In conclusion, there was no dominant spatial pattern in O_3 concentrations along a north-south gradient; rather, the data were influenced by the proximity of the site to large urban complexes. Median O_3 concentrations appeared higher at remote elevated locations, such as Dickey Ridge to the west of the study area. Indeed, MCCP data indicate that high-elevation sites in the Appalachians were exposed to O_3 concentrations similar to those for Dickey Ridge.

Table 5. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the seven AIRS sites in the southcentral United States (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Dickey Rdg	Lewisburg	Hanover	Wake Forest	Davie	Johnston	Cumberland
May	45-55 (98)	24-59 (77)	20-60 (120)	15-61 (104)	20-55 (93)	19-58 (90)	12-62 (110)
June	45-60 (100)	24-61 (95)	20-70 (120)	12-62 (107)	13-58 (96)	14-63 (96)	13-64 (105)
July	49-60 (110)	17-50 (91)	15-70 (135)	12-68 (127)	5-57 (99)	13-64 (110)	5-63 (114)
August	45-60 (105)	14-60 (98)	15-65 (115)	9-64 (133)	5-58 (109)	5-63 (103)	5-58 (108)
September	40-50 (95)	16-48 (76)	10-55 (120)	6-59 (104)	5-54 (100)	11-56 (103)	5-56 (101)
October	30-40 (83)	17-38 (65)	10-40 (95)	5-43 (91)	5-49 (93)	5-47 (91)	5-47 (87)
% hours (May-Oct) ≥ 80 ppb	-	2.1	5.9	4.2	2.1	3.4	3.5

Table 6. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the four AIRS sites in the southeastern United States (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Greenville	Rockdale	Mt. Pleasant	Muscogee
May	11-60 (112)	5-60 (120)	0-47 (88)	5-55 (100)
June	13-60 (107)	5-66 (165)	0-38 (86)	5-55 (97)
July	10-65 (112)	5-67 (170)	0-47 (79)	5-55 (105)
August	8-66 (117)	5-60 (157)	5-48 (91)	5-50 (110)
September	6-56 (101)	5-55 (132)	5-43 (67)	5-50 (100)
October	5-43 (87)	5-45 (108)	10-43 (71)	5-45 (85)
% hours (May-Oct) ≥ 80 ppb	3.8	6.0	1.3	1.8

4.3 Eastern Hardwoods

Eastern hardwood species were studied on FIA plots in Minnesota, Wisconsin, Michigan, Pennsylvania, and Vermont. The FIA data cover a broad geographic area and a wide range of site conditions for specific species. Other projects studied sugar maple in the northeastern United States and Canada and hardwoods in the Tennessee Valley. In addition, in the Ohio River Valley, the northern Lake States, and Pennsylvania, forest condition was studied along pollutant deposition gradients (see Figure 23 for the study sites). These gradient studies used the analogous plot method to reduce variation in response due to changes in tree species composition, tree competition, soils, and other site factors. Analogous plots were selected to have a high degree of similarity along the gradients for variables including fire, insects, pathogens, land use, slope, aspect, soil moisture, overstory species composition, overstory basal area, and stand age (Reed et al., in press 1989). Each analogous plot study represents specific subpopulations of eastern hardwood species.

4.3.1 Forest Condition

In a historical review of hardwood mortality events in the eastern United States, Millers et al. (1989) reported that while many hardwood species have experienced individual declines or mortality events during the last century due to weather, silvicultural practices, disease, or insects, there was no consistent evidence for an association between patterns of mortality and regional atmospheric pollution. Most early studies probably did not specifically examine or describe existing pollution distributions. One exception is a study by McClenahen (1978), in which a reduction in diversity was found in hardwood forests as proximity to industrial air pollution sources increased along a 50-km gradient in the Ohio River Valley.

The Ohio Corridor gradient study investigated possible effects on oak-hickory forest along a four-state pollutant deposition gradient from northwest Arkansas to southern Ohio (LeBlanc et al., 1988). Seven study sites were investigated, with eight permanent plots at each site. The plots were selected to represent narrowly analogous soils and stand characteristics in the oak-hickory type on upper, generally south-facing slopes with low-base soils derived from sandstone and shale. Mortality rates, growth trends, canopy condition, understory condition, fine root production, and soil invertebrate populations were measured. Only data for mortality, growth, and canopy condition have been analyzed as yet. Preliminary results indicate that black oak mortality rates have doubled recently (from 0.7% to 1.2% per year) and were higher in the eastern end of the gradient (Loucks and LeBlanc, 1988). The changes in mortality rates have not been tested for statistical significance.

In other research along the Ohio Valley gradient, LeBlanc (1989) analyzed individual tree basal area trends to assess changes in growth patterns of white oak and black oak between the two periods from 1934-1960 and 1960-1987. Thirty dominant and codominant trees of each species were randomly selected at each of the seven sites and were cored along two radii. Growth decline was determined for each tree separately, and it was defined as mean periodic basal area increment from 1960-1987 that was significantly less than mean basal area increment for the period 1934-1960 ($p = 0.05$). Up to 33% of black oak and 20% of white oak trees exhibited growth decline at sites with very low soil Ca:Al ratios in the B horizon (Loucks and Somers, 1989), while no white oak and only 5% to 10% of black oak exhibited decline at a site with higher Ca:Al ratios. There were no relationships between growth and either tree age, size, or competition, as would be expected due to the criteria used in analogous plot selection.

LeBlanc and Robertson (1989) compared the growth rates of white and black oak over time and the relationship between the growth rates of these two species diverged over time, especially for sites with low Ca:Al ratios (LeBlanc, 1989). LeBlanc and Robertson (1989) also found that

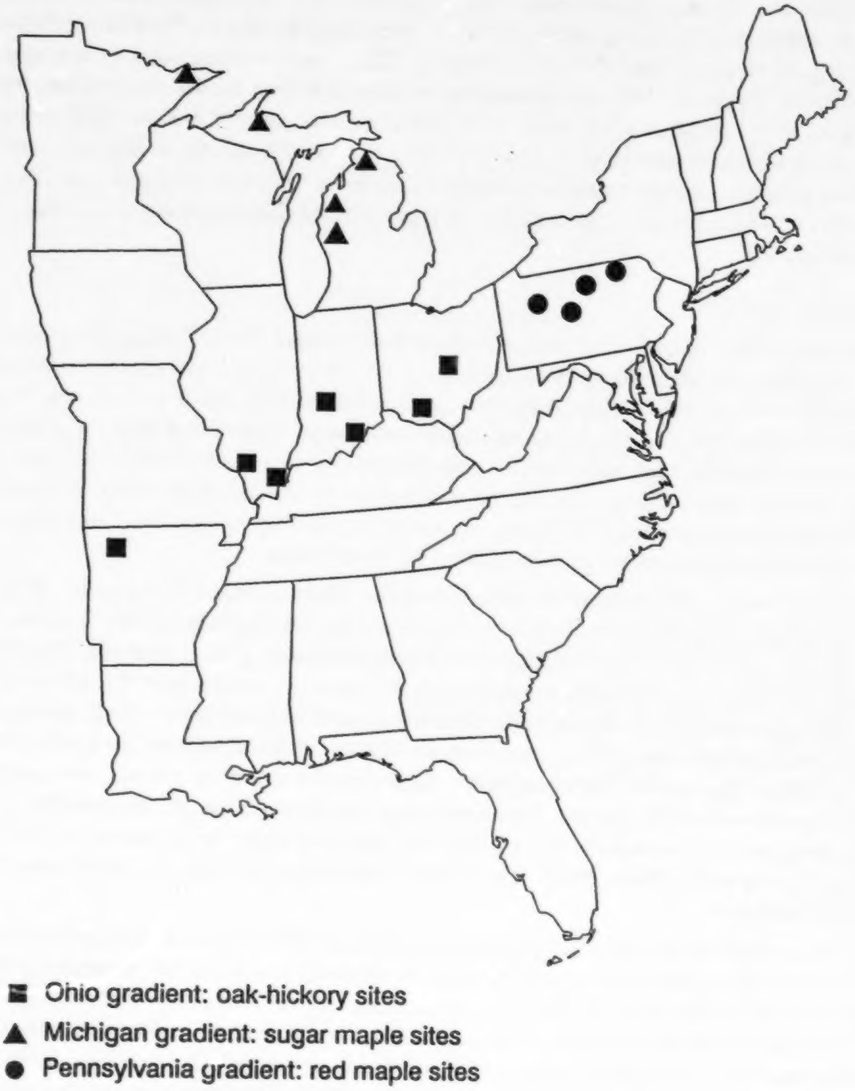
■ Ohio gradient: oak-hickory sites
▲ Michigan gradient: sugar maple sites
● Pennsylvania gradient: red maple sites

Figure 23. Study site locations for the three studies established in eastern hardwood forests along deposition gradients.

between-species growth correlations at sites with low Ca:Al ratios decreased after 1960, with apparently only limited recovery. They concluded that decreased interspecies growth correlations suggest an alteration of common responses to interannual fluctuations of regional climatic conditions. This change might be due to any of several factors (e.g., drought, insect outbreaks, ice storms, silvicultural practices). The authors suggested that reduced interspecies growth correlations, low soil Ca:Al ratios, and increased incidence of individual tree growth decline of black oak may be interrelated.

Loucks and Somers (1989) reported trends of increasing soil acidity, increasing soil S, and increasing soil carbon (C) that followed the trend of increasing atmospheric S deposition across a gradient from Arkansas to Ohio (see Figure 24). However, when they reviewed soil survey results obtained over the same gradient during the 1960s and 1970s, the authors found little indication of the same west-to-east pattern of decreasing soil pH or increasing soil C. Soil C data of the early soil surveys appeared to be highly variable and may not be appropriate for these comparisons. The recent trend of increasing soil C reported by the authors is interesting; since most soil S is organically bound, increases in soil S may simply be due to higher soil C in the eastern portions of the gradient where the pH changes appear to have been greatest (from pH 5.0 to pH 3.6 - 4.0). The gradient in S loading, expressed in relation to the sum of cations in the soil, accounts for much of the variation in pH in the A1 horizon soil samples along the gradient ($R^2 = 0.66$).

Witter et al. (1989) reported on preliminary results from a study of the effects of atmospheric deposition on northern hardwood forests in Minnesota and Michigan. The purpose of the study was to determine whether nutrient cycling or growth efficiency are related to a regional gradient in pollutant deposition. Witter et al. selected five sites along a deposition gradient that increased from northwest to southeast. At each site, precipitation, throughfall, and soil solution chemistry samples were taken. Tree and stand measures taken included leaf area, basal area, root growth and root mortality, and tree size (diameter, height, and crown size).

The deposition measurements showed a two-fold increase in SO_4^{2-} deposition along the gradient, ranging from 10 kg/ha/yr in northern Minnesota to approximately 20 kg/ha/yr in southern Michigan (Liechty and Mroz, 1988). Relating deposition to forest condition, Pregitzer et al. (1988) reported that foliar S concentration in sugar maple increased as SO_4^{2-} deposition increased along the gradient. Concentration of S in litter was also positively correlated with SO_4^{2-} deposition. Concentrations of N in litter were similarly related to NO_3^- deposition. Ratios of S:N in litter increased across the gradient, even though N deposition increased; that is, S in litter increased more than N. The elemental S:N ratio in litter increased from 0.13 to 0.18 from north to south, and the molar S:N ratio increased from 0.05 to 0.08. Witter et al. (1989) suggested that this result was due to pollutant deposition.

Witter et al. (1989) found no differences in standing sugar maple biomass or basal area across the five stands surveyed. There were differences across sites in sugar maple growth efficiency (measured as biomass or basal area increment/leaf area), although the differences were not in any consistent pattern. Growth measures (basal area and biomass increment) were not directly related to deposition levels, but soil chemical characteristics associated with growth were negatively correlated with deposition. Witter et al. concluded that while S, N, and cation nutrient cycles have been influenced by atmospheric deposition, there is no evidence that this change has resulted in decreased productivity. The authors also indicated that since detecting such changes in two years is highly unlikely and that more definitive information on nutrient cycling will require more long-term monitoring.

Ohmann et al. (1988) analyzed 171 remeasured permanent FIA plots distributed across Minnesota, Wisconsin, and Michigan. Wet SO_4^{2-} deposition increased from west to east (northwest

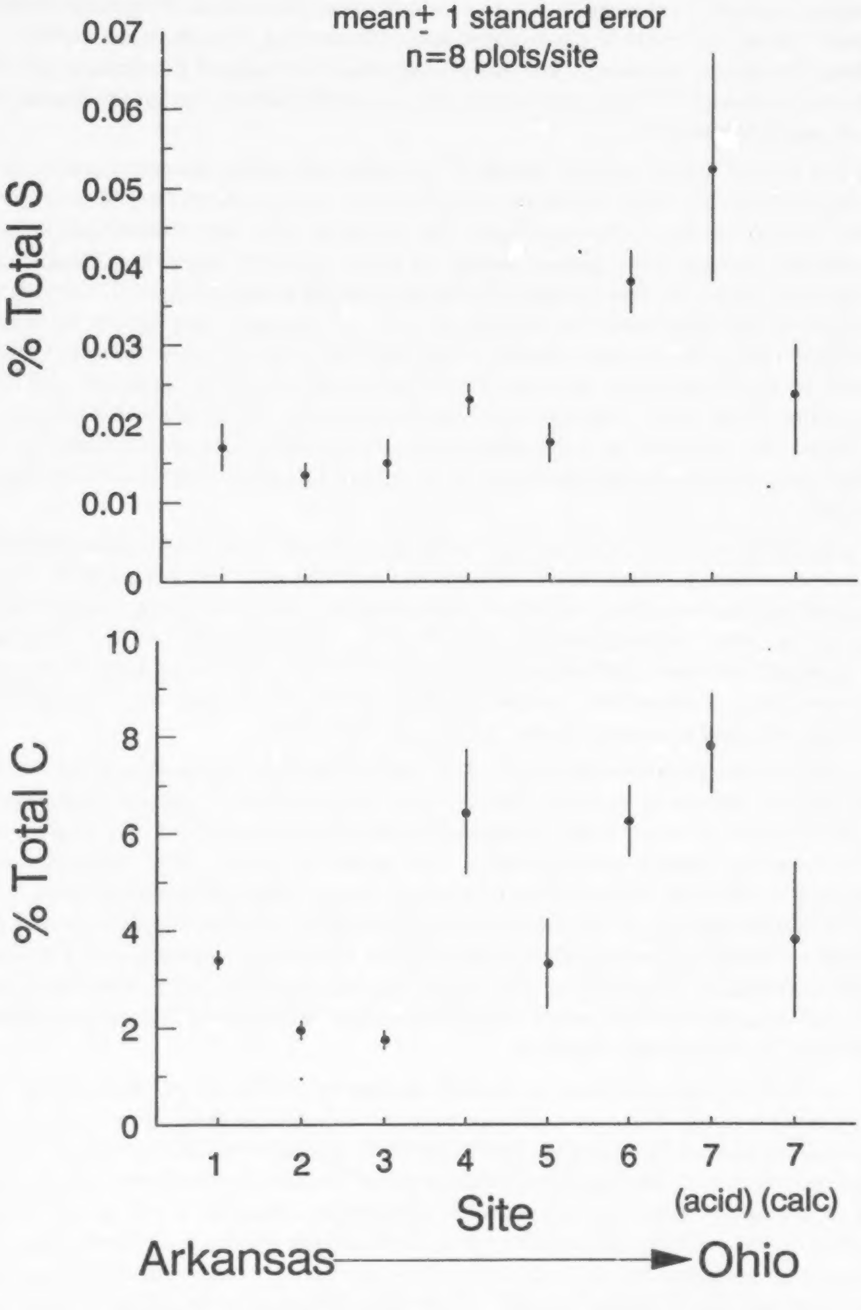

Figure 24. Percent total soil C and S in analogous sites along the Ohio gradient from west to east. Site 7 is displayed on acid soils (4 plots) and calcareous soils (4 plots). The plots on calcareous soils are not comparable to the remaining sites (redrawn from Loucks and Somers, 1989).

Minnesota to southern Michigan). The principal hypotheses tested in the study were that the wet SO_4^{2-} deposition gradient may be correlated with radial increment of trees, and further, that the amount of S in the litter of the forest floor, the mineral soils, and the woody tissues of trees would reflect the gradient. From west to east, estimated SO_4^{2-} deposition ranged from 156 to 380 eq/ha/yr across the gradient. Concentrations of C, N, and S in the forest floor increased from west to east. The Lakes States regional growth model STEMS was used to remove the effects of tree, stand, and site condition. After further adjusting for climate, diameter increments of two hardwood tree species (sugar maple and aspen) were positively correlated with estimated S deposition. Diameter increments of two conifer species (jack and red pine) were negatively correlated with S deposition. These results suggest that S deposition may be influencing forest condition in the Lake States. An additional factor not addressed in this study is the growth reduction that has occurred due to jack pine budworm defoliation during the same period. The eastern region of the study experienced more outbreaks than did the western region. Both jack pine and red pine are hosts of the budworm (I. Millers, personal communication).

Grigal and Ohmann (1989) studied soils along the gradient of increasing atmospheric deposition from Minnesota to Michigan. They found increases in concentrations of Pb and cadmium (Cd) in the forest floor of 171 FIA plots along the gradient (see Figure 25 for the distribution of plots). No trends were observed for Zn, copper (Cu), and nickel (Ni); Ca, Mg, K, P, and sodium (Na) concentrations decreased. The authors speculated that the observed trend in cations was a function of western, wind-borne, soil-derived sources, while the trend in Pb and Cd was due to anthropogenic sources.

David et al. (1988) examined both the soil and the forest floor for C, S, and N concentrations in the plots used by Grigal and Ohmann (Figure 26). David et al. found no differences in S and N concentration of the soils, and a weak trend of increasing S and N concentrations in the forest floor from west to east. When S was adjusted for N concentrations, the trends of S in the forest floor became stronger, and a weak trend of increasing S along the gradient appeared in the soil (Figure 26). Since less than 2% of total S in the soils is inorganic (M. David, personal communication), compounds that form the organic matter in the soils are enriched in S relative to N in the most eastern part of the gradient compared with the western end.

In a study investigating forest responses to atmospheric deposition along a wet deposition gradient, Davis and Skelly (1989) found that lichen species richness was greater on the eastern, low-SO_4^{2-} end of a deposition gradient in central Pennsylvania. They also noted that small woody vegetation species richness was greater at the high-SO_4^{2-} end of the gradient.

Brooks (1989) investigated relationships between forest growth and atmospheric deposition. This study was initiated because Pennsylvania has a general west-to-east (high-to-low) wet SO_4^{2-} and NO_3^- deposition gradient. Although deposition data prior to 1982 were unavailable, the current pattern of deposition has presumably existed for some years. For 200 forest inventory plots in western and central Pennsylvania, Brooks compared periodic annual basal area increment for two growth periods (1960s-1970s and 1970s-1985). The average length of the first and second remeasurement periods were 12.9 and 8.7 years, respectively. Forest stands at the plot locations are of natural origin, with no evidence of harvesting over the last 25 years.

In addition to remeasurement of all 200 plots in 1985, climate information and defoliation history were obtained. Plot defoliation histories were obtained from maps of annual insect defoliations produced by the Pennsylvania Bureau of Forestry. Historical climate records were used to characterize annual soil-moisture conditions (Karl et al., 1983). Climate data were obtained from 10 Climatic Divisions in Pennsylvania, and the climate data for each plot were taken from the division within which the plot was located. Estimates of SO_4^{2-} and NO_3^- deposition were provided by the Institute for Land and Water Research at Pennsylvania State University (Lynch, personal

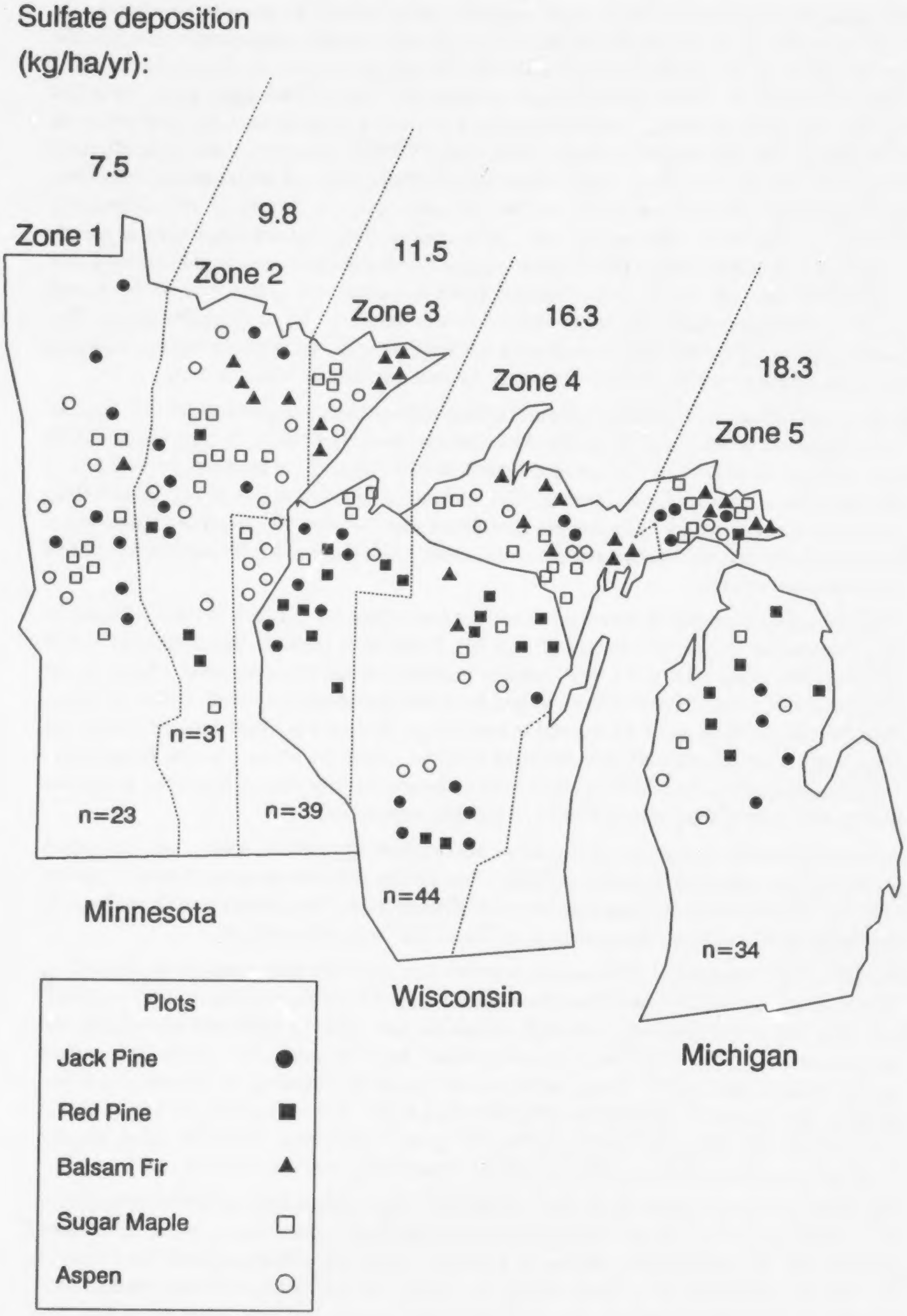
Sulfate deposition
(kg/ha/yr):
7.5
9.8
11.5
16.3
18.3
Zone 1
Zone 2
Zone 3
Zone 4
Zone 5
n=23
n=31
n=39
n=44
n=34
Minnesota
Wisconsin
Michigan
Plots
Jack Pine ●
Red Pine ■
Balsam Fir ▲
Sugar Maple □
Aspen ○

Figure 25. Deposition of SO_4^{2-} on FIA plots used by Grigal and Ohmann (1989) and by David et al., (redrawn from David et al., 1988).

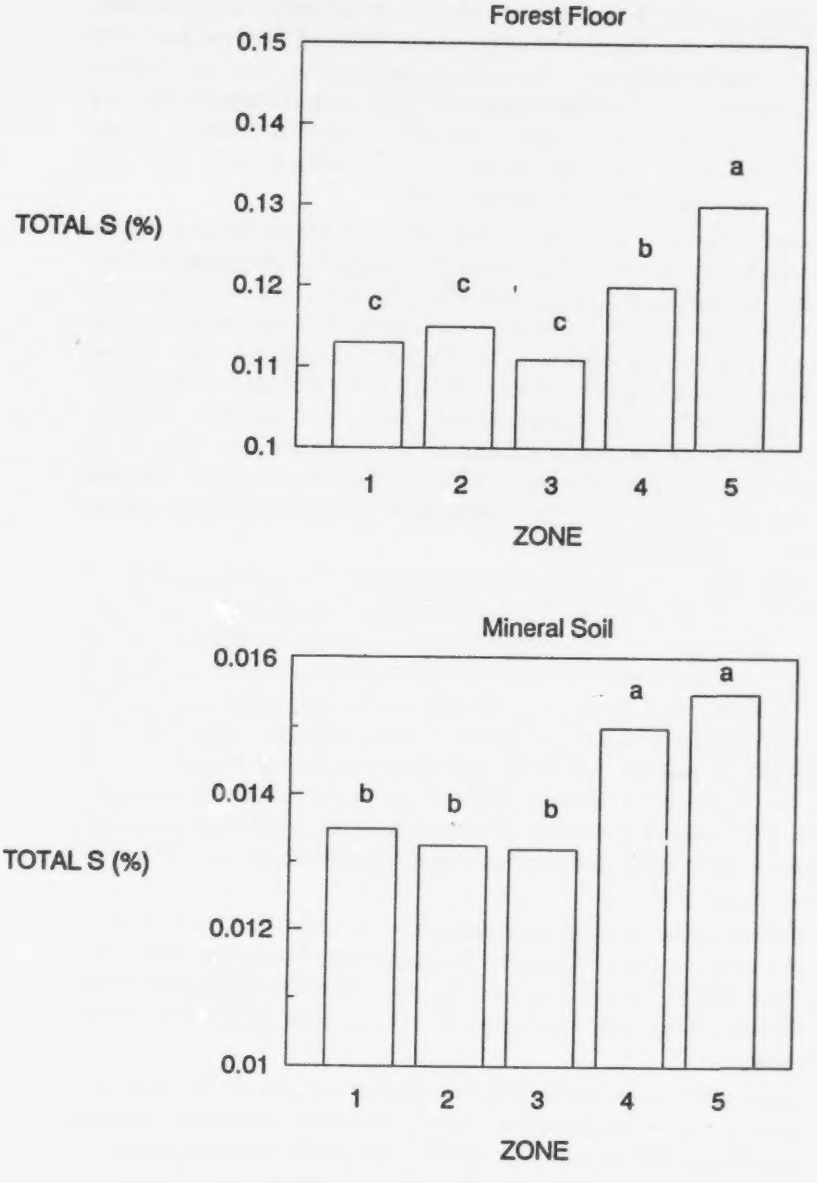

Figure 26. Total S concentrations, adjusted for N concentrations, in forest floor and mineral soils for zones shown in Figure 25. Means with the same letter are not significantly different (n = 169 plots; redrawn from David et al., 1988).

communication). Estimates for 1982-1985 were calculated for each plot on a quarterly basis (January-March, etc.) using multiquadratic equations (Grimm, Institute for Land and Water Research, personal communication). Deposition of SO_4^{2-} ranged from 28.3 to 38.6 kg/ha/yr, and NO_3^- deposition ranged from 20.0 to 24.8 kg/ha/yr.

A preliminary review of the plot data did not show evidence of a broad decline in growth rates. Average periodic annual basal area increment of survivor trees increased from the first to the second growth period. Average periodic annual radial increment for all survivor trees declined by 0.033 cm/tree/yr from the first to the second growth period. This reduction is expected because the plot ages cluster tightly around 70 years, and Pennsylvania forests are essentially second growth, having originated during the early 1900s (Marquis, 1975,1983; Ward, 1983). The radial increment decline therefore probably reflects forest maturation.

Brooks (1989) regressed plot periodic annual basal area increment and mortality for all species and for individual species for the two growth periods against species and competing stocking, stand diameter, drought, defoliation, A-horizon base cations, soil depth, evidence of mottling, and terrain location. Wet SO_4^{2-} and NO_3^- deposition were regressed against residual growth variation for the second growth period. Measures of stand and stocking density were the best predictors of growth variation for both growth periods. Drought was significant in both periods, and defoliation was significant for the second growth period. Neither SO_4^{2-} nor NO_3^- deposition were significant predictors of residual growth variation in any of the analyses. Overall, the regression accounted for only 14% of growth variation in period 2 and 8% in period 1. Analyses of plot-level periodic mortality yielded similar results, with no indication of a significant relationship between mortality and atmospheric deposition.

Barnard and Scott (1988) compared the basal area growth rates of 19 tree species in Vermont for 1965-1972 with growth rates for 1973-1983. The study was conducted to identify species that may be experiencing growth declines. A total of 2,387 trees were measured on 435 FIA plots that represented a variety of forest conditions. Only live trees that were measured on all three survey dates were included in the analysis. The hardwoods tested were red maple, sugar maple, yellow birch, paper birch, American beech, ash, hophornbeam, bigtooth aspen, quaking aspen, black cherry, and northern red oak. Barnard and Scott found no decreases in growth rates for the second time period for any hardwood species. Eight of the 11 hardwood species showed increased growth rates for the second time period. Previous tree diameter and basal area growth rate were the only explanatory variable; factors such as competition, climate, and site (soils, slope, and elevation) were not investigated.

Wu and Groton (1989) analyzed permanent plot remeasurement data from three sites within the Tennessee Valley region in Tennessee and Kentucky. Two sites showed increases in total basal area over time, and a site in Wayne County in southwest Tennessee showed a significant decrease in total basal area (9%) from 1979 to 1987 due to mortality. No cause for this observed change in basal area was apparent.

In a related study, Smith (1990) performed dendrochronological analyses on four hardwood species (white oak, chestnut oak, northern red oak and yellow-poplar). Cores were collected from dominant and codominant trees at each of the 270 TVA permanent plots for up to a total of 15 trees per plot. Cores from the four species represented tree-ring series from 1,843 trees on a variety of sites. Analysis consisted of standardization using the first differences of the logarithms of annual increment, followed by grouping of individual series using multivariate statistical techniques. The relationship between climatic variables and annual increment was then modeled using a time-variant response function. Results of this analysis indicated that, in general, individual trees within a forest stand tend to become less sensitive to climate over time. However, in a majority of the trees studied, a recent trend towards increasing sensitivity to climate was observed that began approximately 10 to 30 years ago. No coincident changes in climate or stand structure occurred that may have influenced this change. At the Wayne County site, many of the tree-ring series showed little to no decrease in sensitivity to climate over the period observed (1895-1986).

The North American Sugar Maple Decline Project is a joint research effort between the United States and Canada (Allen and Barnett, 1989). The purpose of the project is to evaluate rates of change in sugar maple crown condition over three years and to assess whether changes are associated with atmospheric deposition or stand management. Although it is too early in the project to identify rates of change, Allen and Barnett (1989) reported on crown condition data from 1988. At the end of 1988, 166 clusters of plots (five plots per cluster) were established in the United States from Wisconsin to the east coast and in Canada from Ontario to the east coast. Plots were not established in areas of severe damage. Based on first-year data, crown condition of sugar bushes and forest stands had not changed and appeared to be healthy, with 89% of the sugar bushes and 92% of the forest trees showing less than 10% crown dieback. Exceptions are western Massachusetts and southern Vermont, where crown damage due to pear thrips occurred in 1988. A severe drought occurred in Wisconsin, Michigan, and Ontario in 1988, and below-normal precipitation continued in Ontario in 1989. In addition, forest tent caterpillar defoliation occurred in part of the Ontario plots (I. Millers, personal communication). The effects of these events on sugar maple are as yet unknown. In general, the trees showed little damage, and there was no difference between forest stands and managed sugar bushes.

Linzon (1987) reported results of extensive ground and aerial surveys of sugar maple in Ontario. Declining trees were observed throughout the hardwood forests of Ontario on both acidic and calcareous soils. No correlation was apparent between an index of hardwood decline and annual wet SO_4^{2-} deposition for five regions of Ontario.

Summary

In summary, research on eastern hardwoods indicates no widespread changes in tree growth and mortality. However, preliminary results from the Ohio Corridor gradient study indicate that up to 33% of black oaks and 20% of white oaks on sites with low soil Ca:Al ratios in the B horizon show reductions in growth rates that are outside historical patterns. One study found that increased sugar maple growth was weakly correlated with SO_4^{2-} deposition, and another study found that sugar maple growth had not changed. A study in Pennsylvania found no relationship between plot growth or mortality and wet deposition. From northeastern Minnesota to southwestern Michigan, along an increasing gradient of SO_4^{2-} deposition, increases in the amount of S and N cycling have occurred in foliage and litterfall. Concentrations of S in the forest floor and soil increased from west to east along a similar SO_4^{2-} gradient in another survey from western Minnesota to Michigan. In the Ohio Valley, a general increase in total soil S concentrations and soil C concentrations was observed in surveys conducted in 1987-1988.

4.3.2 Wet Deposition

Upper Midwest

Spatial variability in the 1985-1987 precipitation deposition of H^+, SO_4^{2-}, NH_4^+, and in upper midwestern states is presented in Figure 27. There was a trend of higher deposition of all four ions in Michigan and southern Wisconsin compared with Minnesota. The deposition of all four ions was approximately 100 eq/ha/yr in Minnesota and northern Wisconsin. At the Michigan sites, SO_4^{2-} and H^+ wet deposition were about 300 to 500 eq/ha/yr, and NH_4^+ and NO_3^- wet deposition were 200 to 300 eq/ha/yr. Precipitation pH was typically 4.5 to 5.1 in the Midwest, with SO_4^{2-} concentrations between 1 and 2 mg/L. Figure 27 suggests that variability between sites

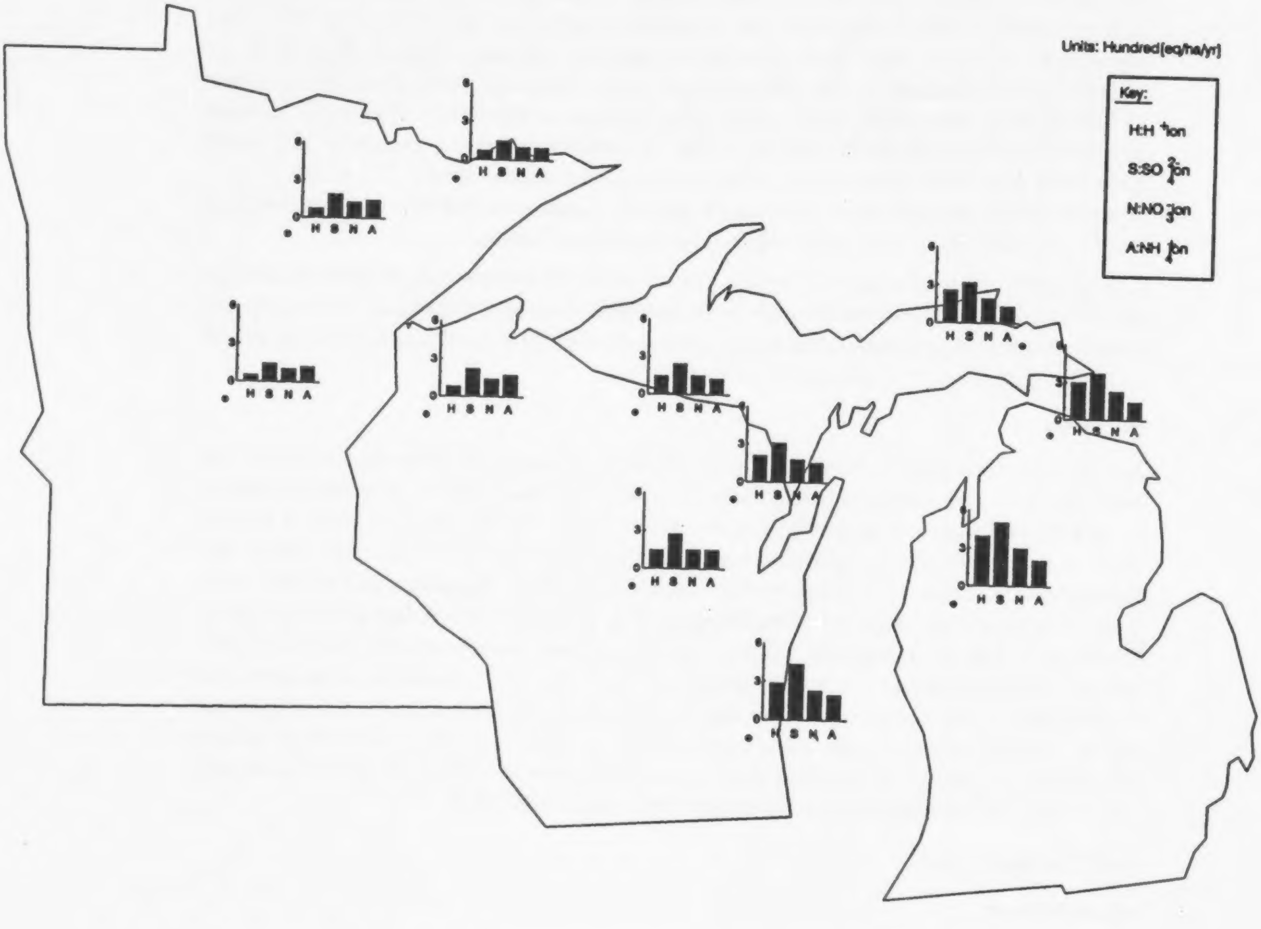
Units: Hundred[eq/ha/yr]
Key:
H:H $^+$Ion
S:SO$_4^{2-}$Ion
N:NO$_3^-$Ion
A:NH$_4^+$Ion
0
3
6
H S N A

Figure 27. Atmospheric wet deposition via precipitation for upper midwestern NADP sites, 1985-1987.

in close proximity (e.g., in northern Michigan) was small compared with differences between sites farther apart (e.g., Minnesota and Michigan).

Lower Midwest

Studies of deposition patterns along a transect from northwest Arkansas to southeastern Ohio have shown differences in both space and time (Armentano and Loucks, 1989), as shown in Figure 28. During 1985 to 1987, deposition in southern Ohio, Indiana, and Illinois was among the highest in the nation. Deposition of SO_4^{2-} and H^+ ions exceeded 500 eq/ha/yr over this region, and in some areas, where local point sources are strong, deposition exceeded 600 eq/ha/yr. Deposition of NO_3^- in these areas ranged from 200 to 250 eq/ha/yr, while NH_4^+ levels were about half that amount. For 1985 to 1987, the only strong gradient in SO_4^{2-}, NO_3^-, or H^+ deposition in the region was from northwestern Arkansas to southern Illinois. There was very little change in the deposition of these ions from southern Illinois to Ohio.

Inferred deposition patterns prior to 1979 indicate that SO_4^{2-} and NO_3^- levels were elevated above background at least as early as the 1950s (Junge and Werby, 1958). Thus, over the central and eastern Ohio corridor area, oak-hickory forests appear to have been exposed to elevated SO_4^{2-} deposition for at least three decades.

Lynch et al. (1984, 1985, 1986, 1987) reported on spatial and temporal variation in atmospheric deposition in Pennsylvania. Data from collection sites supported by the Pennsylvania Department of Environmental Resources, NADP/NTN, and the NPS were analyzed to develop isopleth and three-dimensional maps of annual concentration and deposition levels for various chemical species. The isopleth maps showed significant deposition and concentration gradients across Pennsylvania increasing from east to west. Temporally, neither the SO_4^{2-} nor NO_3^- concentrations exhibited any significant changes over a six-year sampling period. However, there was evidence of increasing NO_3^- deposition and decreasing SO_4^{2-} deposition over the same period.

4.3.3 Ozone

Upper Midwest

The concentration of O_3 in forests of Minnesota, Wisconsin, and Michigan was assessed using data from six AIRS county sites: Sherburne, MN; Dickinson, MN; Washington, MN; Buffalo, WI; Juneau, WI; and Marathon, WI. These sites were chosen to investigate gradients in O_3 exposure from northwest to southeast, west to east, and north to south (see Table 3 and Figure 2). The results in Table 7 indicate that:

1) O_3 concentrations increased towards the southcentral portion of the region, where O_3^{med} concentrations were greater than 60 ppb, and, on occasion, exceeded 80 ppb during May through October. Extreme concentrations (O_3^{worst}) of up to 150 ppb have been recorded at Buffalo. Elsewhere, O_3^{med} rarely exceeded 40 ppb, with daily maxima less than 100 ppb.

2) all sites exhibited O_3 levels greater than 20 ppb during the night.

3) all sites exhibited little diurnal and seasonal fluctuations.

4) across the region, Buffalo and Juneau were the only sites with recorded hourly concentrations greater than 120 ppb. However, these sites were only operational for one year (1979 for Juneau and 1981 for Buffalo).

In conclusion, forests to the southern portion of the growth trend area near Buffalo were exposed to very high O_3 concentrations. Elsewhere, exposures were generally lower, although few concentrations exceeded 120 ppb.

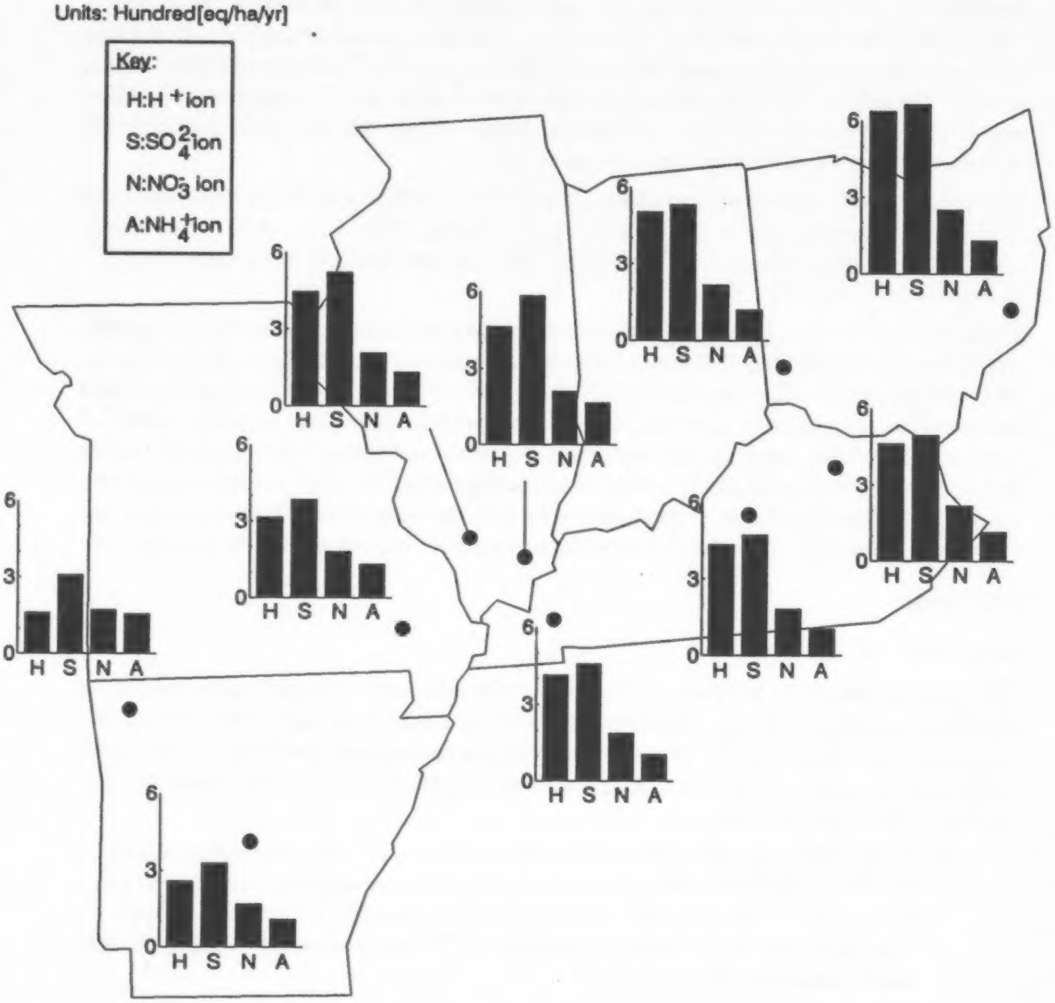
Units: Hundred[eq/ha/yr]
Key:
H:H + ion
S:SO$_4^{2-}$ ion
N:NO$_3^-$ ion
A:NH$_4^+$ ion
6
3
0
H S N A

Figure 28. Atmospheric wet deposition via precipitation for lower midwestern NADP sites, 1985-1987.

Table 7. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the six AIRS sites in the upper Midwest (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Sherburne	Washington	Buffalo	Juneau	Marathon	Dickinson
May	19-45 (95)	23-46 (94)	71-106 (143)	22-53 (86)	27-53 (83)	26-48 (75)
June	11-40 (74)	20-43 (91)	58-98 (141)	31-59 (118)	20-50 (94)	26-44 (80)
July	10-42 (100)	18-43 (110)	55-90 (149)	8-55 (98)	15-47 (75)	18-40 (75)
August	10-38 (90)	17-40 (110)	51-95 (137)	14-45 (63)	15-44 (83)	17-37 (82)
September	8-28 (58)	12-31 (79)	40-69 (106)	6-33 (61)	14-37 (86)	16-34 (70)
October	2-12 (20)	16-38 (60)	35-56 (86)	- (-)	19-34 (57)	20-33 (59)
% hours (May-Oct) ≥ 80 ppb	0.4	0.6	0.1	3.3	3.3	3.3

Lower Midwest

In Pennsylvania, an ozone gradient existed during the 1988 and 1989 growing seasons, with the actual levels of ozone at each monitoring site being higher in 1988 than in 1989 (Skelly et al., 1989). This gradient coincided with the SO_4^{2-} and NO_3^- gradient in Pennsylvania.

Several growth trend studies were performed along a north-east / south-west transect through southern Ohio, Indiana, Illinois, and Arkansas. There are several ozone monitors across this region, of which a subset is used here to illustrate patterns in O_3 concentrations (see Table 3 and Figure 2); data were from EPRI (Duncan Falls, 1979, and Rockport, 1979) and the AIRS database (North Little Rock, 1979-1987). The results shown in Table 8 indicate that:

1) on average, the three sites had similar O_3^{med} concentrations, with median hourly concentrations ranging between 20 and 60 ppb.
2) forests near the North Little Rock site in the southern portion of the transect were exposed more frequently to O_3 concentrations greater than 120 ppb than those close to the other two sites. Despite this, all three sites experienced a similar percent of hours with concentrations greater than 80 ppb.
3) all sites experienced elevated concentrations (> 30 ppb) during nighttime hours.

In conclusion, forests across this transect were probably exposed to similar average O_3 concentrations. Forests in the southern portion were exposed to higher maximum concentrations than elsewhere.

4.4 Western Conifers

In the FRP, western conifer species have been studied in four areas. Ponderosa pine was studied in central and southern Arizona, the Sierra Nevada of California, and the Front Range of the Colorado Rocky Mountains. Douglas-fir was investigated in the Front Range, in Arizona, and in the Puget Sound region of Washington. These studies were dendrochronological: trees at least 50 years old were studied to determine if growth rates have reduced in recent years. The study sites represented areas with high, moderate, and low pollutant exposures. Inferences of tree populations in the West are limited by the relatively small number of species and site conditions sampled.

4.4.1 Forest Condition

Chronic symptoms of O_3 damage to ponderosa and Jeffrey pine in the San Bernardino Mountains of southern California have been documented (Miller et al., 1989). Forests north of the San Bernardino Mountains, in the southern Sierra Nevada, are also subject to relatively high levels of O_3. To investigate possible recent changes in tree growth that may be associated with increased exposure to pollutants, Peterson and Arbaugh (1988) analyzed increment cores of ponderosa pine from 56 mixed conifer forest sites. Eight sites were chosen in each of seven federal administrative units (five National Forests and two National Parks); the sites had differing management histories and resulting stand characteristics as well as recorded symptomatic foliar injury data. In each unit, four sites were classified as symptomatic and four as asymptomatic. Symptomatic sites were those for which there was evidence of exposure to air-borne pollutants (O_3) from urban centers in western California. The symptomatic sites were thus on the western edge of the Sierra Nevada, and the asymptomatic sites in the interior.

Sampled stands were from mixed conifer forests with ponderosa pines greater than 50 years old. Ages of the sampled trees varied; most were over 80 years old. National Park stands have been almost undisturbed in this century. Thus, the stands tend to be of mixed age classes and to reflect the effects of disturbances such as fire and mortality of older trees. Most National Forests,

Table 8. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the three AIRS sites within growth trend areas in the lower Midwest (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Duncan Falls	Rockport	NorthLittle Rock
May	19-61(92)	22-57(88)	27-52(129)
June	28-68(100)	25-68(99)	21-54(121)
July	17-61(108)	18-54(92)	23-61(134)
August	18-59(78)	9-54(81)	20-59(118)
September	10-42(60)	7-57(93)	0-52(106)
October	15-54(74)	11-40(83)	4-42(98)
% hours (May-Oct) ≥ 80 ppb	3.6	2.6	3.5

however, have been subject to timber harvest and silvicultural treatments. Almost all sites in several of the National Forests had at least some prior thinning.

Using basal-area increment rather than ring width, a variant of the Kalman filter representation using autoregressive modeling was applied to the basal area increments for each tree. A growth trend was identified as a consistent increase or decrease in the estimated slope and intercept over 10 or more years. The date of a change in the trend was recorded to the appropriate decade and the proportions of increase and decrease per decade for each site were calculated. Differences between the two proportions were calculated for 1951-1980, the period assumed to represent potential pollutant effects. A normal distribution was fitted to the data for asymptomatic sites that did not exhibit abnormal releases from management practices. The 5th and 95th percentiles of this distribution were then used to judge whether the remaining sites exhibited changes. Using the longest available record of precipitation for the Sierra Nevada (Yosemite NP), differences in annual precipitation from the century average were calculated; the differences were then averaged in decades and correlated with the adjusted growth increases and decreases for each site since 1910.

The increases and decreases up through the 1940s were correlated with above- and below-normal precipitation, respectively. From 1951 to 1980, both increases and decreases occurred. There were somewhat more increases (especially since 1961), most noticeably in the asymptomatic sites in the 1960s. This could indicate that symptomatic sites were adversely affected relative to the asymptomatic sites (see Figures 29 and 30). The authors reported that, compared with symptomatic sites, asymptomatic sites generally have shown more growth losses relative to increases since 1950. In most sites of either type, increases exceeded decreases. Correlations between annual precipitation and changes in basal area growth were, with one exception, positive for all sites showing growth increases, and inconsistent for sites with growth decreases. While precipitation trends may account for many of the growth changes, non-climatic factors may have resulted in decreases in at least some of the sites.

In a further study of the same data, D.L. Peterson et al. (1989) reported that O_3 injury to ponderosa pine needles was most severe in the southern region and least severe in the north. This finding corresponds with the south-to-north (high-to-low) gradient in O_3 exposure, and it corroborates data from crown condition surveys of the USFS and the NPS (California Air Quality Data, 1987). The injury in the south extends north for approximately 250 km. The highest levels of injury were measured in Sequoia-Kings Canyon National Park. Most two-year-old needles had some chlorosis, with 50% of these needles showing 6% to 25% chlorosis. More than 50% of one-year-old needles had chlorotic injury as well. Because symptomatic sites in the south showed a general trend of growth decreases since 1950 in an area with high levels of O_3 exposure and injury, Peterson et al. suggested that O_3 injury has played a role in growth reductions in this area.

In a study of both ponderosa pine and Douglas-fir in Arizona and New Mexico, Graybill and Rose (1988) constructed 41 tree-ring site chronologies. Most sites were ponderosa pine (21) and Douglas-fir (11), and the rest were pinyon pine, southwestern white pine, and white fir. Twenty of the collections were in the southern sector of the region where pollution has been greatest. Several southern sites were chosen because of their proximity to copper smelters and to Phoenix and Tucson, the primary sources of pollutants. The stands varied in slope, exposure, drainage, and openness, and settings ranged from the forest interior to near the forest border. At least two samples were taken from 10 to 20 or more trees per stand, when possible. Older trees were sampled to provide a relatively long time series of natural growth variation prior to the influence of pollutants. The range of initial dates was from 1361 to 1778, with a median of 1620.

Ring widths were measured to the nearest 0.01 mm, and each dated sequence was standardized, typically by fitting a simple negative exponential or straight line. The individual indexed sequen-

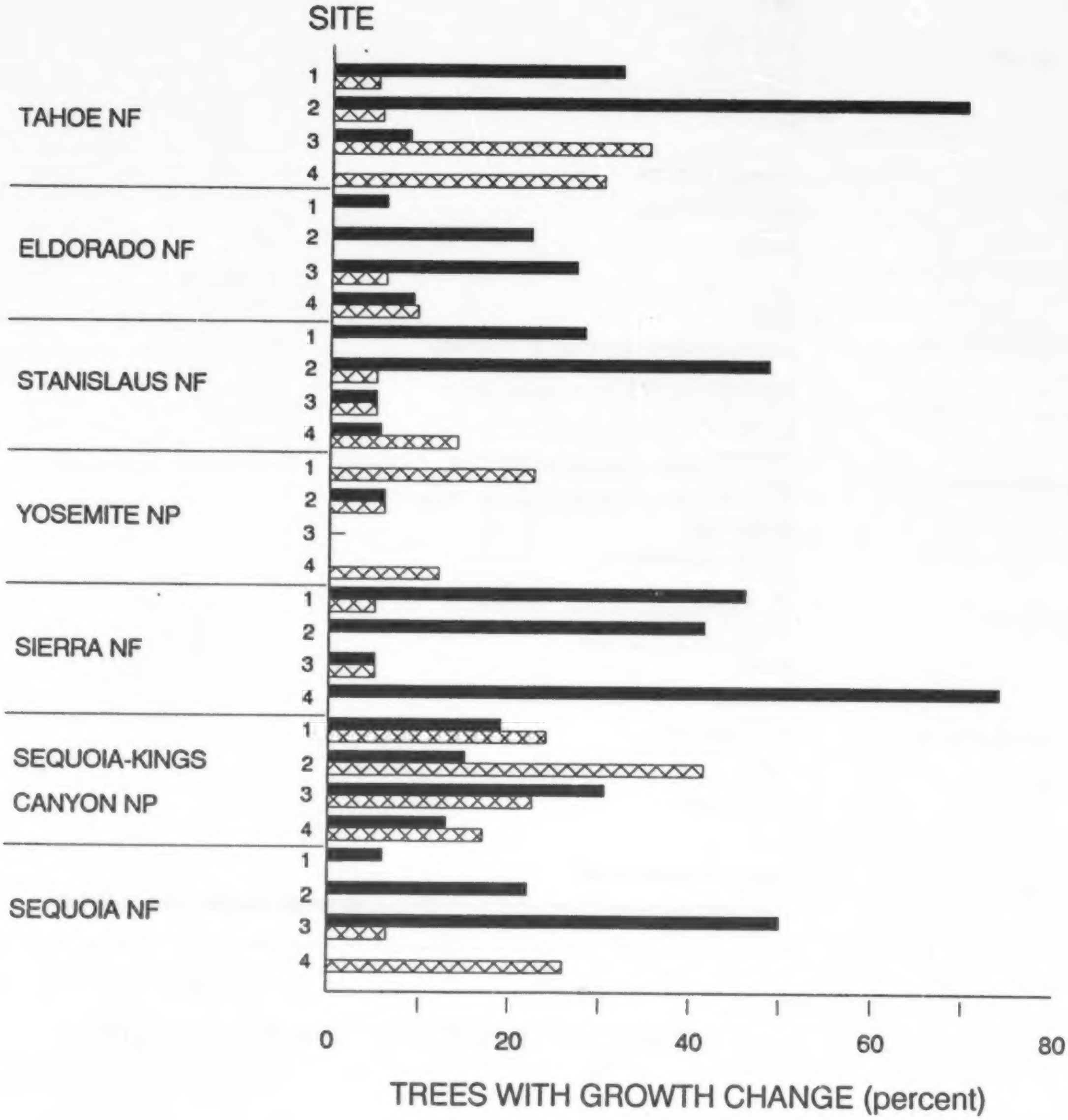

Figure 29. Percent of trees in each symptomatic site with growth decreases (hatched bar) and increases (solid bar) since 1950 as determined by Kalman filter analysis (redrawn from Peterson and Arbaugh, 1988).

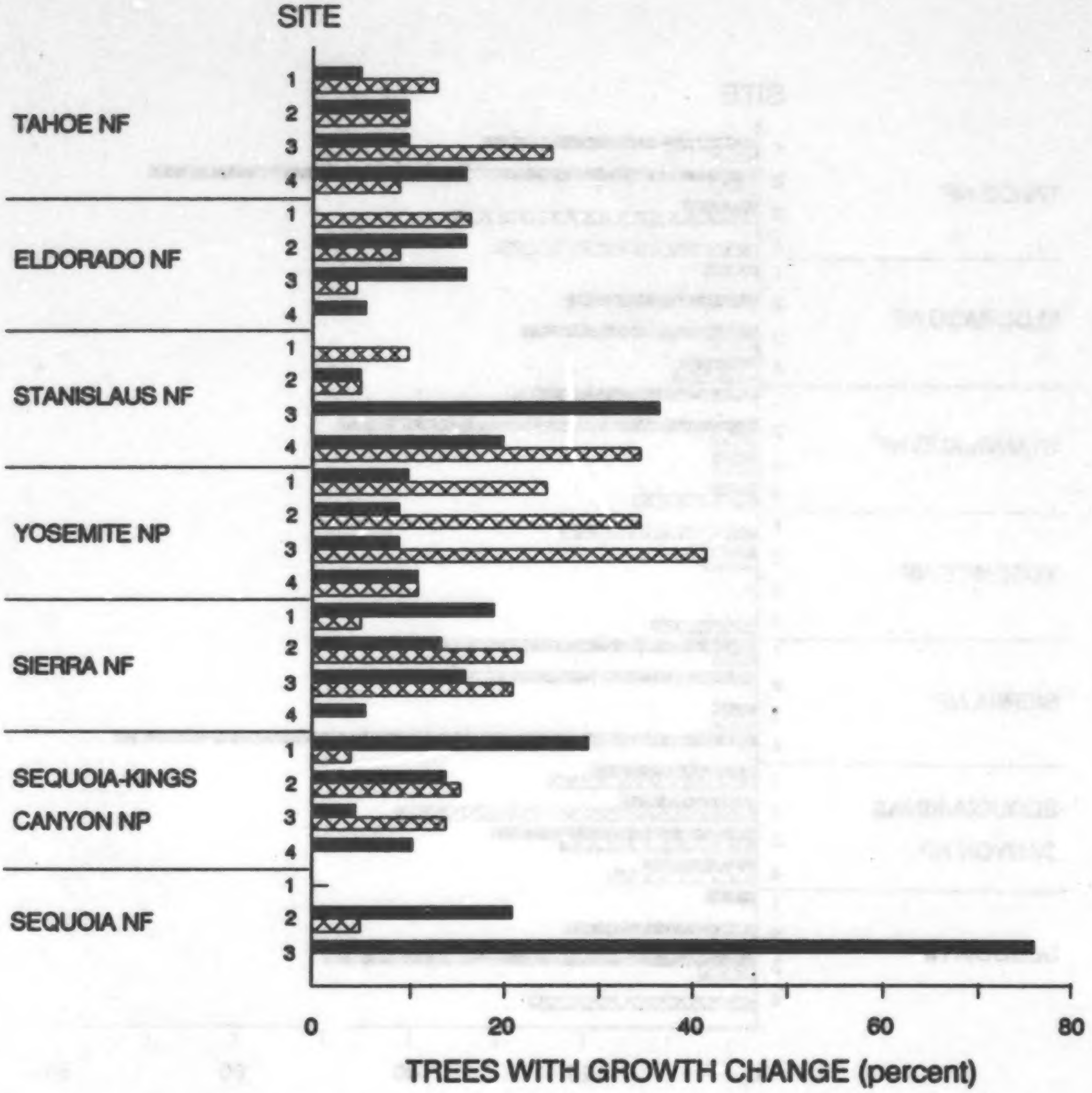

Figure 30. Percent of trees in each asymptomatic site with growth decreases (hatched bar) and increases (solid bar) since 1950 as determined by Kalman filter analysis (redrawn from Peterson and Arbaugh, 1988).

ces were averaged, and ARMA modeling was then applied to these site chronologies. These series were then used to predict tree growth from climate for two baseline periods: 1897-1950 and 1897-1960. Ultimately one climate variable was used: total precipitation from August of the year prior to, through July of the year of ring formation ($p < 0.05$).

Each baseline period was divided into two halves; the first half was used for calibration (parameter estimation) and the second for verification, and then vice versa. The final calibration used data from the full baseline periods. The relationships from these two baseline periods were then used to predict tree growth for 1951-1986 and 1961-1986, respectively. For 1951-1986, 36 of the 41 r^2 values were significant, and for 1961-1986, 35 r^2 values were significant ($p < 0.05$). An F test indicated that in 24 of the 41 cases for 1951-1986 (and in 20 cases for 1961-1986), actual and projected variances were unequal. For the first period, 22 of these cases had an actual variance that was greater than projected; this was to be expected since in none of the calibrations was all of the variance accounted for by regression on climate data.

There is evidence of some relative reduction in growth rate for at least a segment of the forest area studied, although there were some spatial differences in actual and predicted growth levels. Across all species, 63% (1951-1986) to 58% (1961-1986) of the southern series showed mean growth decreases ($p < 0.05$). In the north, 38% showed growth decreases from 1951-1986, and 24% for 1961-1986. A Chi-squared test of the proportions was significant for 1961-1986 ($p < 0.05$), but not for 1951-1986; this is suggestive but inconclusive evidence of growth disparity in the two regions.

North-south differences by species were also reported. For 1951-1986, 75% of southern ponderosa pine chronologies showed less growth than expected, compared with 46% in the north. For 1961-1986, these values were 63% in the south and 23% in the north. For Douglas-fir, 71% of chronologies in the south, but only 25% in the north had less growth than expected for 1951-1986. Sites located near pollution sources showed no consistent growth reduction.

Graybill and Rose evaluated all of the ponderosa pine for foliar damage and other morphological characteristics indicative of stress, and a substantial number of cores were taken for elemental analysis. These additional analyses are not complete. However, ponderosa pine outside of the Tuscon area exhibited no foliar damage clearly attributable to pollution. In the Rincon Mountains just east of Tuscon, 11 ponderosa pine trees (23%) showed some O_3 damage, but only three appeared to have growth reductions in recent years.

Graybill and Rose concluded by suggesting three alternative but not necessarily mutually exclusive hypotheses for some of the observed growth decreases. The first hypothesis is that lack of growth response to relatively wet conditions of the past several decades following a severe mid-century drought may be related to different genetic tolerances. Second, local pollutant deposition may have affected the climate-tree growth relationship, especially in the south. Third, natural biological competition may be responsible for some growth decreases. Wet conditions in the early 20th century may have encouraged the establishment of many trees that, during the next 30 to 60 years, may have competed with older trees for water, nutrients, and light.

Douglas-fir was studied in the Puget Sound area by Brubaker et al. (1989). They obtained increment cores from old-growth trees at 12 sites in the Puget Sound area, and one site in the H.J. Andrews Experimental Forest, OR. Three of the Puget Sound sites were in the Olympic Mountains, a region deemed to be of pristine air quality. The nine remaining sites were in the Cascade Mountains, in areas potentially exposed to pollutants from the Vancouver-Portland corridor. Nearly all sites were in stands dominated by Douglas-fir at least 450 years old. Usually two cores were taken from 20 to 30 trees at each site.

Brubaker et al. removed the natural growth trend by fitting a straight line to logarithms of the ring widths. Sequences of stand averages were formed from the residuals, and these were then smoothed using a robust locally weighted regression procedure. The smoothed sequences were used to generate another set of residuals that were assumed to contain the random, exogenous, and endogenous disturbances (including, perhaps, pollution). These individual sequences were smoothed to take out disturbance effects that should be principally endogenous and leave the residual random year-to-year variation. The primary purpose of this step was to remove the effect of competition. Cross correlations of the resulting "white-noise" sequences were formed for all pairs of trees at a given site.

Three climate variables were used to predict growth: mean winter temperature (January-April), summer wetness (precipitation, June-August) and winter dryness (precipitation, November-May). Two regressions were computed: one on mean winter temperature and the other on summer wetness and winter dryness. The calibration period was from the start of the climate series data (generally prior to 1900) to 1945. The verification period was 1945-1960, and the decline period was from 1960 to the end of the sequence. Fits were made to individual tree sequences.

Sites in both the Olympics and Cascades showed a consistent trend: growth increased from 1880 to 1950, leveled off or slightly declined until 1975, and then increased. The overall growth increase since 1880 is probably related to a regional temperature increase. The long-term trend in the data is consistent with patterns found previously in a variety of forest zones in the Cascades. Growth losses since 1960 outnumbered growth increases in both site averages and the consistency of trends across trees within a site. At 5 of the 12 sites, the regression model fit the post-1960 data less well than the pre-1960 data. However, sites showing a poorer fit after 1960 did not show consistent relationships with other measures that may suggest a pollution effect. For example, sites with a poorer fit to post-1960 data did not consistently show growth decreases after 1960 as might be expected if pollution were causing changes in growth. Thus, no patterns appeared to be related to pollution sources.

Peterson and Arbaugh (1989) conducted a dendroecological study of ponderosa pine in the Front Range of the Colorado Rocky Mountains to: 1) quantify stand and regional growth trends and natural variability; 2) compare recent with previous growth trends, as well as sites with different pollutant levels; and 3) determine the effects of pollution exposure, climate, and other environmental factors on growth. Twenty "exposed" sites, assumed to experience high concentrations of air pollutants, were selected on eastern slopes facing urban areas, and 10 "protected" sites were selected on western slopes. Stands had a closed canopy with ponderosa pine dominant and were at least 50 years old. At each stand, two cores were extracted at dbh from 24 trees. Height, dominance based on crown classification, crown condition, and length of sapwood and heartwood of cores were also recorded. Group variability was determined for each site by principal components analysis. The Kalman filter was also used to determine individual tree growth trends. Regional patterns and temporal trends in estimated parameters were evaluated subjectively based on stand dynamics, climate trends, and other environmental factors.

Since O_3 is the most likely pollutant affecting tree growth in the Front Range, O_3 data were collected from the NPS in Lakewood, CO, and a monitoring station was constructed at Manitou Experimental Forest near protected sites. Ozone concentrations were higher in the central part of the range than in the north or south. The highest levels were recorded in valley areas during summer months, several times exceeding 100 ppb at each site. Mountain sites experienced only moderately high O_3 levels. Levels at Manitou were higher than expected, sometimes exceeding Colorado Springs values, which questions the assumption that this site is protected.

No visible injury due to O_3 was found, which corroborates previous studies of this area (Stolte, personal communication, unpublished data from Rocky Mountain National Park). Needle retention did not differ among sites. Although O_3 levels in the Front Range were not as high as other regions where ponderosa pine shows visible injury, such as the Los Angeles Basin and the southern Sierra Nevada, O_3 was at levels associated with chlorosis in these other areas. The authors suggested that the Rocky Mountain variety of ponderosa pine may be more resistant to O_3 injury than the Pacific variety.

Growth trends were reported to be regional in scale. Reductions in growth were attributed to low precipitation in the 1940s and to aging. A growth increase in the 1980s was attributed to high precipitation. No growth trends were found that could be associated with temporal or spatial O_3 exposure patterns. The authors concluded that it is unlikely that any growth effects are due to elevated O_3 levels. The effects of climate and other environmental factors on tree growth have not yet been fully evaluated.

Graybill (1989) reported on a broad survey initiated in Colorado in 1988 to determine if there was any change in the tree growth-climate relationship that could be related to pollutant deposition. Increment cores were taken from trees at 20 sites just west of the urban corridor from Fort Collins to Colorado Springs. The sample included 153 ponderosa pine, 51 limber pine, 44 Engelmann spruce, 42 Douglas-fir, and 34 lodgepole pine. Site elevation ranged from approximately 1700 m to 3500 m. Significant relationships between divisional precipitation data and detrended tree growth were developed during the baseline period (pre-1950). For almost all of the chronologies, the amount of unexplained variance remained stable between the calibration and verification periods, indicating that the climate-growth relationship remained about the same.

The 16 chronologies from low elevations (below 2600 m) showed increased growth in response to precipitation of the current spring and summer and late summer of the prior year. The chronologies at higher elevations showed diverse growth responses. Results from t-tests indicated no differences between actual and predicted growth in all 16 cases, although a Wilcoxon test suggested that mean differences were significant in 5 of 16 cases. A review of actual and predicted values of radial growth did not indicate any strong divergent growth for these five cases. The results did not suggest any widespread change in climate-tree growth relationships since 1950. There were not adequate data to characterize either the amount or types of pollutants that may affect forest growth.

Summary

Ponderosa pine foliar injury consistent with that typical of O_3 exposure occurs over a 250-km corridor in the southern Sierra Nevada of California. Because sites exposed to increased levels of O_3 have shown a general trend of a reduction in the number of growth increases since 1950 along with increased foliar injury, O_3 has been suggested as a possible cause. Some foliar injury to ponderosa pine needles has occurred in the Rincon Mountains of southern Arizona. No visible injury to ponderosa pine was evident in Colorado, nor has the growth-climate relationship changed. In the Puget Sound region, radial growth of Douglas-fir has generally increased, and this increase is probably related to a regional temperature increase.

4.4.2 Wet Deposition

The limited NADP/NTN precipitation chemistry data for western forests, shown in Figures 31 and 32, allow only preliminary conclusions about spatial variation. Precipitation deposition for 1985-1987 was substantially lower in the West (with the exception of California) than in the East. Wet deposition of H^+, SO_4^{2-}, and NO_3^- was usually below 100 eq/ha/yr at western NADP/NTN forest sites except in California. The NADP (1988) reported precipitation pH above 5.0 for the West, with SO_4^{2-} concentrations between 0.5 and 1 mg/L. Concentrations of H^+, SO_4^{2-}, and NO_3^- are expected to be higher (lower pH) near urban areas such as Los Angeles or point sources

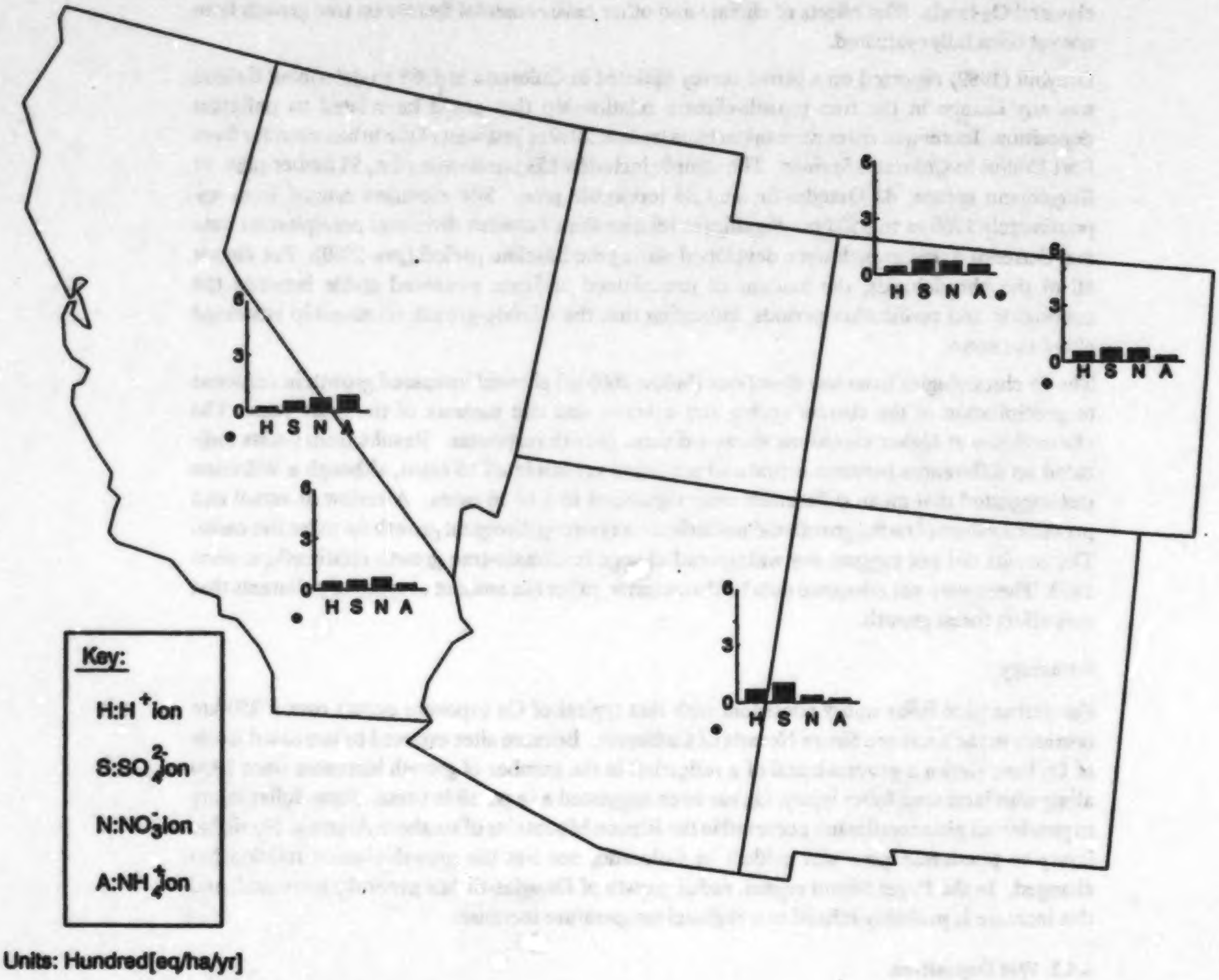
6
3
0
H S N A
6
3
0
H S N A
6
3
0
H S N A
6
3
0
H S N A
6
3
0
H S N A
Key:
H:H+ ion
S:SO4 2- ion
N:NO3- ion
A:NH4+ ion
Units: Hundred[eq/ha/yr]

Figure 31. Atmospheric deposition via precipitation for southwestern NADP sites, 1985-1987.

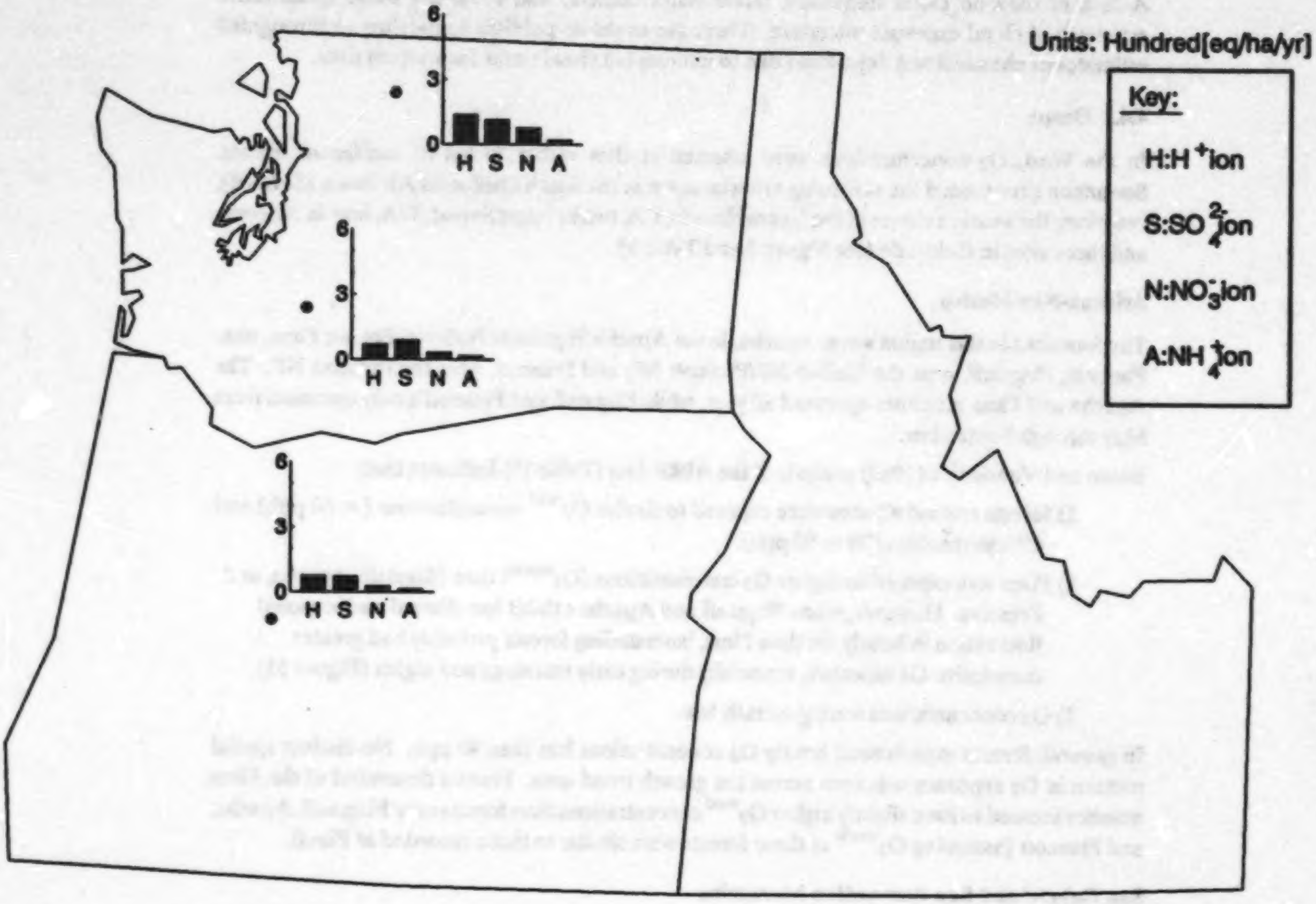
Units: Hundred[eq/ha/yr]
Key:
H:H$^+$ion
S:SO$_4^{2-}$ion
N:NO$_3^-$ion
A:NH$_4^+$ion
6
3
0
H S N A
6
3
0
H S N A
6
3
0
H S N A

Figure 32. Atmospheric deposition via precipitation for northwestern NADP sites, 1985-1987.

such as smelters and coal-fired power plants (Liljestrand and Morgan, 1981; Vong et al., 1988) than the pH indicated by NADP contour maps. Blanchard et al. (1989) reported SO_4^{2-} deposition via precipitation of 200 to 500 eq/ha/yr in the Sierra Nevada Mountains of California while NO_3^- deposition varied from 300 to 700 eq/ha/yr depending on site location. The Sierra Nevada differ from most sites in that NO_3^- deposition exceeds SO_4^{2-} wet deposition.

Regarding elevational variation, cloud water was more acidic than precipitation was at western sites, including a remote site on the Washington coast (Basabe et al., 1989) and a site in the Colorado Rockies (Miller and Borys, 1988). A limited amount of data summarized by Muir and Böhm (1989) suggests that the chemical exposure of western forests increases above cloud base. A lack of data on cloud frequency, liquid water content, and chemistry make quantitative estimates of cloud exposure uncertain. There are as yet no published modeling or throughfall estimates of chemical wet deposition due to intercepted cloud water for western data.

4.4.3 Ozone

In the West, O_3 concentrations were assessed at sites within 20 km of coniferous forests. Seventeen sites passed the screening criteria: seven in the South California Air Basin (SoCAB), two along the western slopes of the Sierra Nevada, CA, two in Puget Sound, WA, four in Arizona, and three sites in Colorado (see Figure 2 and Table 9).

Arizona-New Mexico

The four sites in this region were: Apache, in the Apache Sitgreaves National Forest; Pima, near Phoenix; Flagstaff, near the Kaibab NF/Prescott NF; and Prescott, near the Prescott NF. The Apache and Pima monitors operated all year, while Flagstaff and Prescott's only operated from May through September.

Böhm and Vandetta's (1989) analysis of the AIRS data (Table 10) indicates that:

1) forests around all sites were exposed to similar O_3^{med} concentrations (< 60 ppb) and 95th percentiles (70 to 90 ppb).
2) Pima was exposed to higher O_3 concentrations (O_3^{worst}) than Flagstaff, Apache, and Prescott. However, since Flagstaff and Apache exhibit less diurnal and seasonal fluctuation in hourly O_3 than Pima, surrounding forests probably had greater cumulative O_3 exposure, especially during early mornings and nights (Figure 33).
3) O_3 concentrations were generally low.

In general, forests experienced hourly O_3 concentrations less than 90 ppb. No distinct spatial pattern in O_3 exposure was seen across the growth trend area. Forests downwind of the Pima monitor seemed to have slightly higher O_3^{med} concentrations than forests near Flagstaff, Apache, and Prescott (assuming O_3^{worst} at these forests were similar to those recorded at Pima).

San Gabriel and San Bernardino Mountains

The San Gabriel and San Bernardino Mountains form a barrier between the SoCAB and the San Joaquin Valley to the north and the Mojave Desert to the east. Photochemical smog in the SoCAB is well documented (e.g., Barry and Chorley, 1976). However, air quality in the surrounding mountains is poorly documented (Bytnerowicz et al., 1987), despite evidence of foliar injury on ponderosa and Jeffrey pine in the San Bernardino Mountains east of the SoCAB (Miller, 1983).

Miller et al. (1986) showed that in the San Bernardino Forest for 1974-1978:

1) there were high-to-low gradients in O_3 concentration from west to east, and from north to south.

Table 9. Site information for AIRS ozone sites in the western United States. Numbers refer to Figure 2.

Number	Site Name	Latitude	Longitude	Elevation (msl)	AIRS Description	summary period
1	Santa Barbara	34° 27.7' N	120° 01.7' W	25	remote	Jan 80-Dec 87
2	Ventura County	34° 45.0' N	119° 05.2' W	1600	remote	Jan 80-Jul 84
3	Ojai	34° 26.8' N	119° 16.2' W	233	suburban-mobile	Jan 84-Dec 87
4	Burbank	34° 10.9' N	118° 18.5' W	170	suburban-industrial	Jan 80-Dec 87
5	Azusa	34° 08.1' N	117° 55.4' W	185	suburban-industrial	Jan 80-May 88
6	Lake Gregory	34° 14.6' N	117° 16.5' W	1397	rural-near urban	Jan 80-Dec 87
7	San Bernardino	34° 06.4' N	117° 16.5' W	320	city center-commercial	Dec 80-May 86
8	Sequoia NP	36° 30.0' N	118° 50.0' W	526	rural	Jun 82-Sep 85
9	Fresno County	37° 06.7' N	119° 18.7' W	1723	remote	Jan 80-Dec 83
10	Yreka	41° 43.7' N	122° 38.1' W	809	suburban-residential	Mar 81-Dec 87
11	Apache	33° 45.0' N	109° 00.0' W	2462		Jan 80-Jan 84
12	Pima	32° 17.4' N	111° 00.0' W	695	suburban-residential	Jan 80-Sep 88
13	Flagstaff	35° 12.8' N	111° 36.7' W	2117	suburban-commercial	May 80-Sep 85
14	Prescott	34° 33.5' N	112° 28.9' W	1673	suburban-commercial	Apr 81-Sep 84
15	Boulder County	40° 0.8' N	105° 16.0' W	1635	city center-resident	Jan 81-Sep 88
16	Denver	39° 45.1' N	104° 59.2' W	1591	city center-commercial	Jan 80-Jun 88
17	Colorado Springs	38° 49.4' N	104° 49.4' W	1842	city center-commercial	Jan 80-Jun 88
18	Cedar River	46° 23.2' N	121° 57.9' W	210	rural	Apr 86-Sep 88
19	King County	47° 33.1' N	122° 2.7' W	22	rural	Jan 80-Jun 88

Table 10. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the four AIRS sites in Arizona (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Flagstaff	Prescott	Pima County	Apache Sitgreaves
May	40-60 (80)	20-50 (70)	10-60 (90)	50-50 (90)
June	40-60 (90)	20-50 (80)	10-60 (100)	50-50 (90)
July	30-50 (70)	10-40 (70)	10-60 (100)	40-50 (90)
August	30-50 (80)	10-50 (70)	10-60 (110)	40-50 (70)
September	30-50 (70)	10-40 (60)	10-50 (120)	30-40 (50)
October	- (-)	- (-)	0-50 (90)	40-40 (50)
% hours				
(May-Oct)				
≥ 80 ppb	0.18	0.09	0.77	0.14
≥ 120 ppb	0.00	0.00	0.00	0.00

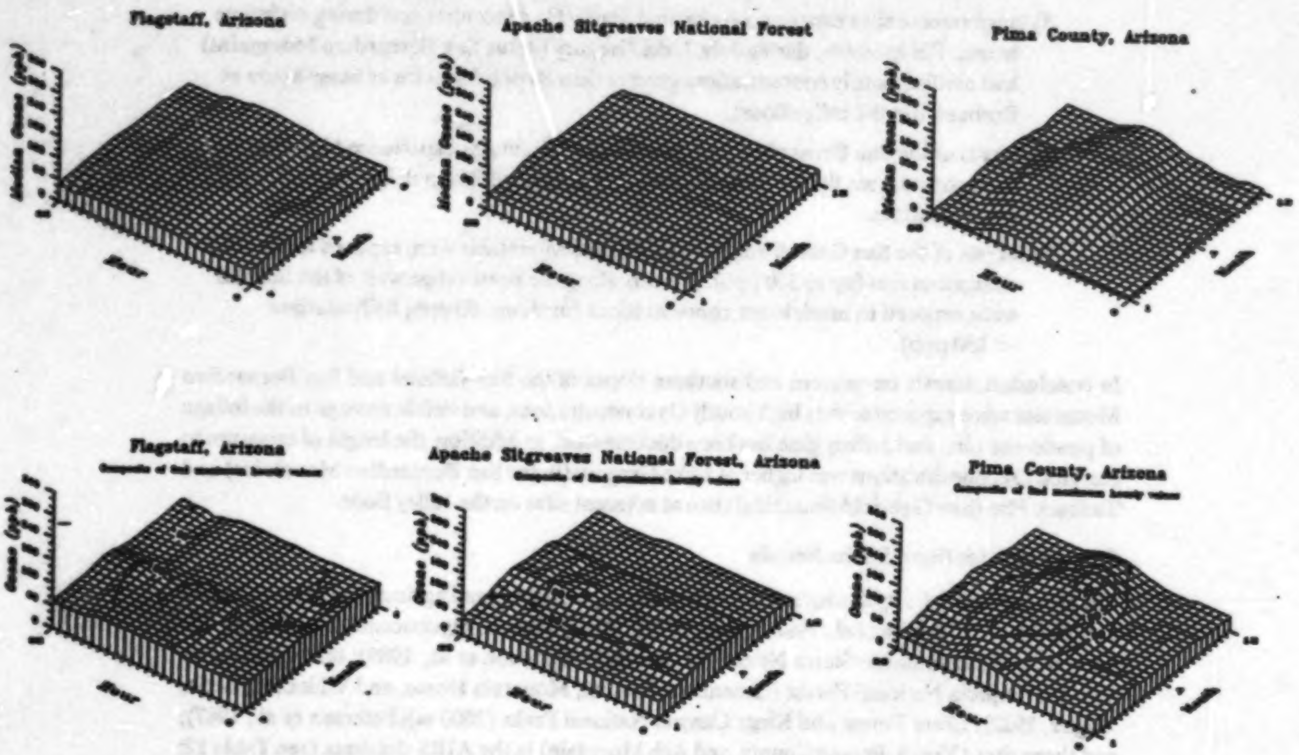

Figure 33. Median and worst-case O_3 surfaces for Apache and Flagstaff (north) and Pima (south), illustrating the higher exposures in the north and the higher peak concentrations in the south.

2) O_3 concentrations during daylight hours did not differ much with elevation, but nighttime concentrations were substantially higher at mountain sites than at sites on the valley floor (approximately 70 ppb in the west and 40 ppb in the east).

3) At two sites in the northwest, 24-hour averages ranged between 90 and 140 ppb, with average maximums between 200 and 240 ppb.

Analysis of AIRS data for 1980-1987 at three sites by Böhm and Vandetta (1989) showed the following (see Table 11):

1) O_3^{med} exposures and frequency of occurrence of hourly O_3 concentrations >80 or 120 ppb were greater in eastern and southeastern portions of the SoCAB than at sites in the west or at high elevations in the San Gabriel Mountains.

2) western sites in the SoCAB showed less diurnal variation in hourly concentrations than sites towards the eastern and southeastern portions of the SoCAB.

3) more remote sites experienced elevated hourly O_3 concentrations during nighttime hours. For example, during July, Lake Gregory (in the San Bernardino Mountains) had median hourly concentrations greater than 80 ppb for twice as many hours as Burbank (on the valley floor).

4) forests of the San Bernardino and San Gabriel Mountains experienced O_3 that was transported from the SoCAB, illustrated by the time delay in daily maximum concentrations.

5) forests of the San Gabriel and San Bernardino Mountains were exposed to very high concentrations (up to 320 ppb). Forests along the coast range west of the SoCAB were exposed to much lower concentrations (medians 60 ppb; daily maxima <150 ppb).

In conclusion, forests on western and southern slopes of the San Gabriel and San Bernardino Mountains were exposed to very high hourly O_3 concentrations, and visible damage to the foliage of ponderosa pine and Jeffrey pine has been documented. In addition, the length of exposure to elevated O_3 concentrations was higher at Lake Gregory (in the San Bernardino Mountains) and Tanbark Flat (San Gabriel Mountains) than at adjacent sites on the valley floor.

Central and Southern Sierra Nevada

The Sierra Nevada lie upwind of several metropolitan, industrial, and agricultural areas (Morgan and Slusser, 1978; Smith et al., 1981; Unger, 1978). Information on O_3 concentrations is available for several sites: southern Sierra Nevada forests (D.C. Peterson et al., 1989); three sites on or near the Sequoia National Forest (Greenhorn Summit, Mountain Home, and Whitaker Forest; Vogler, 1982); Giant Forest and Kings Canyon National Parks (2000 m)(Peterson et al., 1987); and three sites (Yreka, Fresno County, and Ash Mountain) in the AIRS database (see Table 12; Böhm and Vandetta, 1989).

The results from these studies indicate that:

1) there was a general increase in the magnitude of O_3^{med} concentrations and the number of hours with levels 80 or 120 ppb, respectively, towards the southern portion of western Sierra Nevada slopes; Vogler (1982), however, found discontinuities in this pattern due to intrusion of polluted air up canyons near urban settlements such as Bakersfield and Fresno.

2) O_3 season typically lasted from May to mid-October, peaking from June through September. During the rest of the year, hourly concentrations usually did not exceed

Table 11. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the seven AIRS sites in the southern California Air Basin (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Santa Barbara	Ventura	Ojai	Burbank	Azusa	Lake Gregory	San Bernardino
May	20-50 (110)	20-60 (110)	20-70 (140)	0- 80 (260)	10-100 (280)	30-110 (270)	0-110 (260)
June	10-50 (110)	20-60 (120)	20-70 (120)	0-110 (270)	0-140 (330)	40-150 (300)	0-140 (280)
July	10-50 (90)	10-50 (120)	10-70 (120)	0-120 (310)	0-160 (350)	30-170 (320)	0-150 (330)
August	10-50 (90)	10-50 (110)	10-80 (130)	0-130 (290)	0-170 (360)	30-160 (300)	0-150 (320)
September	10-50 (140)	10-50 (140)	10-70 (160)	0-110 (280)	0-120 (370)	20-110 (260)	0-100 (280)
October	10-60 (140)	10-50 (130)	10-60 (120)	0- 70 (260)	0- 80 (370)	10- 60 (200)	0- 60 (190)
% hours (May-Oct)							
≥ 80 ppb	1.22	3.18	7.50	16.64	21.91	33.73	26.82
≥ 120 ppb	0.08	0.17	0.34	6.74	11.31	15.26	13.70

Table 12. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the three AIRS sites along western slopes of the Sierra Nevada, CA (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon. ** refers to the low data capture for the Sequoia NP site.

Month	Yreka	Fresno County	Sequoia NP
May	10-50 (70)	- (90)	30- 70 (100)
June	10-50 (80)	40-60 (100)	- (130)
July	10-50 (70)	40-70 (110)	- (130)
August	10-50 (80)	40-60 (120)	50-100 (140)
September	0-40 (100)	40-60 (100)	40- 80 (130)
October	0-30 (70)	30-50 (90)	- (100)
% hours (May-Oct)			
≥ 80 ppb	0.03	1.15	22.13**
≥ 120 ppb	0.01	0.00	0.30

50 ppb, although values from 100 to 120 ppb were occasionally recorded in April and November.

3) O_3^{med} concentrations during May through October ranged between 30 to 90 ppb, with peak levels of 130 ppb (Ash Mountain), 160 ppb (Whitaker Forest), and 170 ppb (Greenhorn Summit). Daily maxima usually occurred in the late afternoon.

4) sites exhibited slight diurnal fluctuations in concentrations. Vogler (1982) noted that daily maximum hourly averages at mountain sites were frequently equal to, or slightly less than, maxima at adjacent urban sites in the San Joaquin Valley. Böhm and Vandetta (1989) found that evening, night, and early morning O_3 levels were usually elevated in summer months (80 to 100 ppb).

In summary, forests in the southern Sierra Nevada were exposed more frequently to O_3 concentrations of a higher magnitude than forests in the north. Intrusion of O_3 into canyons close to urban settlements created discontinuities in this pattern.

Northwest Washington

Information on O_3 concentrations in forested areas of northwestern Washington is available from the AIRS database (Snohomish and King County), the Washington State Department of Ecology (Olympic National Park), and EPRI (Cedar River site). The hourly O_3 data from these sites were obtained by Böhm and Vandetta (1989) and analyzed with the same procedures described for the AIRS data. The results indicate (Table 13):

1) O_3 concentrations west of Seattle were substantially lower ($O_3^{med} < 20$ ppb; $O_3^{worst} < 60$ ppb) than those in the Puget Sound Basin where median concentrations as high as 50 ppb were calculated and daily maxima in excess of 120 ppb have been recorded.

2) concentrations were higher to the east (Cedar River) and southeast (King County) than to the northeast (Snohomish).

3) highest exposures were found at forests in and around the Cedar River site, where concentrations were elevated at night (> 20 ppb) and where the highest concentrations in the Puget Sound Basin have been recorded.

In conclusion, forests west of Seattle were exposed to low O_3 concentrations. Forests on western slopes of the Cascades, south and southeast of Seattle-Tacoma, were exposed to higher O_3 concentrations than elsewhere in the Basin.

Front Range, Colorado

Although the prevailing winds across the western United States are from west to east, topographic and mesoscale meteorological conditions along the eastern slopes of the Rocky Mountains often induce south-easterly winds during the summer (Mitchell, 1976; Lewis et al., 1984) that may transport pollutants from Denver-Boulder-Fort Collins to forests of the northern Front Range.

Front Range O_3 data are not readily available. All three AIRS sites close to the Front Range are located in the urban centers of Boulder, Denver, and Colorado Springs, as shown in Table 9 (Böhm and Vandetta, 1989). Ozone concentrations are monitored at higher elevations at Manitou in the southern portion of the Front Range, and Glacier Lakes in southern Wyoming.

The results in Table 14 indicate that:

1) sites located at the base of the Front Range experienced O_3^{med} concentrations of less than 60 ppb, although O_3^{worst} ranged between 100 to 160 ppb have been recorded.

2) exposures were greater north of Denver, where O_3^{med} concentrations as high as 60 ppb and O_3^{worst} concentrations of 130 ppb have been recorded.

Table 13. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the four AIRS sites in northwestern Washington (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Olympic NP.	Snohomish	Cedar River	King County
May	10-20 (50)	10-40 (80)	- (80)	0-40 (110)
June	10-20 (50)	10-30 (110)	20-50 (100)	0-40 (110)
July	10-20 (40)	0-30 (110)	20-40 (100)	0-40 (120)
August	10-20 (40)	0-40 (90)	20-50 (120)	0-40 (120)
September	0-10 (30)	0-30 (70)	20-40 (140)	0-30 (80)
October	- (10)	- (60)	20-30 (60)	0-20 (50)
% hours				
(May-Oct)				
≥ 80 ppb	0.00	0.10	1.19	0.33
≥ 120 ppb	0.00	0.00	0.15	0.01

Table 14. Diurnal range in hourly median ozone concentrations and the second highest ozone concentration recorded in history (in parentheses) for the three AIRS sites along the eastern slopes of the Front Range, CO (ppb). Lowest median concentrations usually occurred during the night, highest concentrations in the early to late afternoon.

Month	Boulder County	Denver	Colorado Springs
May	30-50 (110)	1-4 (80)	10-50 (90)
June	30-60 (120)	10-50 (100)	10-50 (90)
July	20-60 (130)	10-50 (120)	10-50 (90)
August	20-60 (130)	0-50 (110)	10-50 (100)
September	20-50 (100)	0-40 (90)	10-40 (70)
October	10-40 (80)	0-30 (60)	0-30 (60)
% hours			
(May-Oct)			
≥ 80 ppb	1.35	0.43	0.08
≥ 120 ppb	0.02	0.03	0.00

3) concentrations were highest in July and August.

In conclusion, forests in the Front Range to the north of Denver are probably exposed to relatively high O_3 levels during upslope wind conditions. Other high-elevation sites in the Rocky Mountains suggest that exposures of approximately 30 ppb are common.

5 SUMMARY AND CONCLUSIONS

As stated earlier, the purpose of this document is to address the following scientific questions:

1) Are changes in forest condition greater than can be attributed to natural variability?
2) What spatial patterns exist in forest condition and how do they relate to spatial patterns of pollutant exposure?

In this section, we summarize the principal findings of the research presented in this document, and we answer these two questions.

5.1 Summary of Principal Findings

The research used to answer the two scientific questions is summarized by forest type.

5.1.1 Northern Red Spruce

Prior to and during the initiation of the FRP, studies of red spruce across the Northeast indicated that increases in mortality at high elevations and radial growth reductions at both high and low elevations had occurred in the mid-1960s. Research suggested a large-scale synchronous decline in red spruce condition that was attributed to acidic deposition. However, before changes in forest condition can be related to deposition levels, the role of natural variability should be evaluated.

Mortality

At low elevations (below 800 m) in the Northeast, red spruce mortality is considered to be within expected levels. The proportion of recent dead red spruce is greater in the Adirondack and Green Mountains than in mountains to the east (30% versus 14%).

For many high-elevation sites in the Northeast, large proportions of standing dead red spruce have been reported since the mid-1960s. The number of standing dead red spruce at high elevations increased between the mid-1960s and mid-1980s in the Adirondack, Green, and White Mountains. At the highest elevations (above 1200 m), 30% to 60% of red spruce basal area was dead. The Adirondack and Green Mountains showed an increased percentage of standing dead red spruce with increasing elevation. This mortality has not yet been associated with any natural agents of mortality, although frequent winter injury events starting in the late 1950s and early 1960s resulted in loss of foliage and buds and may have initiated the period of increased mortality. The greatest mortality has occurred at or above 1000 m, which may represent an extreme in the range of red spruce. Thus, an environment that may already be considered marginal for red spruce could become intolerable for the species if additional stresses were imposed, such as unfavorable climate and/or the addition of acidic deposition.

It is not known whether mortality rates of red spruce at high elevations are outside the range of natural variability; however, it does appear that rates are above normal expectation. Natural variation includes all worst cases in the complete range of mortality episodes that can be attributed to natural phenomena, and therefore this variation can be very large. For mature red spruce, mortality rates of greater than approximately 30% over 25 years would be considered greater than expected by many researchers.

Growth

Red spruce radial growth data indicate a widespread pre-1950 radial growth increase and a subsequent post-1960 growth decrease at low and high elevations throughout the northeastern United States. The percentage of stands that are declining in radial increment in the Northeast is currently unknown because data were aggregated over sites, and because studies were not specifically designed to determine this percentage. In a study in northern Maine, where half of the red spruce population in the United States occurs, an estimated 54% of low-elevation sites sampled showed a pre-1950s increase in radial growth followed by a post-1960s decrease.

Some research suggests that the 1960s reduction in mean radial increment is related to a breakdown of past climate-growth relationships since the early 1960s. Climatic extremes (warm summers and cold winters) may have been an initiating factor at high elevations in the early 1960s. Both young and old trees at high and low elevations reflect this climatic signal.

Other research indicates that the reduction of radial increment at high elevations is accentuated by increases in growth that could have resulted from release of competition in the 1940s and 1950s. Stand dynamics appears to be affecting radial growth rates at low and high elevations. Stand history events such as releases from competition could explain the increases in growth rates preceding growth reductions in the 1960s, and this may be independent of a regional climate signal found in all age classes of trees. Still other research indicates that old trees (over 100 years old) are not declining, and the reduction in mean radial growth since 1960 is proportional to the number of young trees used in calculating the mean. While these results do not rule out the possible impact of acidic deposition in the 1960s growth decline, they suggest that this growth decline should be assessed in light of release history.

Crown Condition

Surveys of red spruce crown condition at a range of elevations in New York, Vermont, New Hampshire, Massachusetts, and West Virginia indicate that crown conditions have deteriorated since 1985. There were no changes in mortality over the same period. In Vermont, crown deterioration was associated with slower radial growth since the early 1960s, and trees with healthy crowns had greater radial increment than trees with less healthy crowns.

5.1.2 Southern Red Spruce

In the southern Appalachians, concern over red spruce and Fraser fir exists because of concentrated areas of standing dead trees and possible growth declines of surviving trees.

Mortality

In the southern Appalachians, Fraser fir mortality caused by an introduced insect, the balsam woolly adelgid, has increased since the early 1960s. In spruce-fir forests of this region, 24% of the total area has more than 70% of dominant and codominant trees standing dead, and 70% of the total area has less than 30% standing dead trees. Areas with the greatest standing dead were dominated by Fraser fir that had been killed by the adelgid. Red spruce standing dead typically averaged less than 10% throughout the region, and recent mortality events have been associated with ice storms. In West Virginia, only 7% of the total land area of spruce type has greater than 10% mortality.

Growth

Since the mid-1960s, radial increment of red spruce at the highest elevations has reduced slightly. Climate, air pollutants, and the balsam woolly adelgid have been suggested as contributors. The effects of past disturbances and the resulting influences of stand dynamics have not been investigated. Radial growth and pollutant exposure have not been correlated because deposition estimates and tree-ring data were obtained from different locations.

Crown Condition

Red spruce crown condition has generally declined from 1985 to 1988. Two ice storms in the winter of 1986-1987 resulted in crown damage to 28% of surviving trees at the highest elevations.

During the same period at lower elevations, the number of trees with more than 90% of their crown intact decreased by 20%. In a survey in West Virginia, 40% of the red spruce trees at elevations above 1000 m had less than a 40% live-crown ratio, with most of the low crown ratios associated with natural crown suppression.

5.1.3 Southern Pines

In the early 1980s, preliminary analyses of forest inventory data indicated that radial growth of southern pines may be decreasing. The possibility that these reductions may be associated with air pollution led to further analysis of survey data.

Growth

Comparisons of the fifth FIA inventory with the third and fourth inventories indicated that average annual radial increment of southern pine species in natural stands decreased by 30% to 50% in the period between the third (1957-1966) and fifth (1977-1985) surveys. In contrast to natural stands, planted stands showed no regional growth reductions. It has been suggested that genetic improvement and fertilization of planted stands may offset potential growth reductions.

For natural stands, attempts to correlate changes in stand structure with individual tree and stand growth have been limited because stand structure effects are not comparable for the time periods of interest. For example, the radial growth reduction of the same diameter classes of trees over time may be expected because the sampled population of southern pine trees changed from young stands with low stand density to old stands with high stand density over the last three decades. Thus, covariance models should be used with caution because adjusted means may require substantial extrapolation of the regression lines to a region where only a few data points exist.

Data from permanent plots established in the early 1930s at Duke Forest were used to estimate growth increment as a function of stand density. Residuals from this model indicated that growth trends in loblolly pine may be cyclic, showing less growth increment during 1965-1984 than from 1934-1965, and increased growth rates since 1984. Because a baseline rate of growth has not been established, abnormally high prior growth cannot be distinguished from abnormally low recent growth. Comparisons of growth rates with climate data suggested no obvious correlation.

In another study that used permanent plots remeasured over 28 years, average basal area growth of loblolly pine over comparable ranges of stand age, site index, and stand density indicated significant reductions in growth for the most recent period (1976-1980). However, a regional ice storm occurred just after the 1959 measurements, and damaged crowns may not have recovered.

A study that compared annual survivor growth from plots distributed throughout the natural range of loblolly pine found that the relationships between basal area growth and plot-level covariates were generally consistent across study sites.

In summary, neither individual tree growth reductions nor decreases in productivity per unit area have been firmly established for southern pines.

5.1.4 Eastern Hardwoods

Eastern hardwood forests are subjected to varying levels of pollutant deposition. Because strong spatial gradients in SO_4^{2-} and NO_3^- deposition are known to exist in this area, research was designed to investigate the possible influence of pollutants on forest health and productivity. Available data are from surveys, literature reviews, and gradient studies. Although there is some information on eastern hardwood crown condition, the data are preliminary and have not been emphasized in this report.

Mortality

In a historical review of hardwood declines, many species were reported to have experienced mortality events during the last century due to weather, silvicultural practices, pathogens, or insects. There was no consistent association between patterns of mortality and regional atmospheric deposition. However, preliminary results from the Ohio Valley indicate that black oak mortality rates have recently doubled from 0.7% to 1.2% per year and are higher in the eastern region of the study, where sulfate deposition is highest. These results have not been tested for statistical significance.

Growth

Some weak associations exist between tree diameter growth and wet deposition in the Great Lakes region (Minnesota to Michigan). After adjusting for climate, diameter growth of two hardwoods (sugar maple and aspen) was positively correlated with estimated sulfur deposition, and diameter growth of two conifers (jack pine and red pine) was negatively correlated with sulfur deposition. A jack pine budworm outbreak that increases in severity along the sulfur deposition gradient may be contributing to decreases in red and jack pine growth rates. In another study along this gradient, sugar maple growth rates in Michigan were not directly related to deposition levels, and thus far there is no evidence of decreased productivity. In a study in the Tennessee Valley region, two of three areas increased in total basal area over time, and a third area in southwest Tennessee showed a decrease in total basal area by 9% from 1979 to 1987 due to mortality at the study plots. No cause for the mortality in this area was apparent. In a study along the Ohio River Valley, one study found that the incidence of reduced radial growth for black oak trees occurred at sites characterized by low soil Ca:Al ratios. However, no direct relationship has been found to date between growth rates and the spatial gradient of acidic deposition. A study in Pennsylvania found no relationship between plot growth or mortality and wet deposition. In Vermont, of 11 hardwood species studied, none showed decreases in growth rates for the most recent time period (1973-1983), and eight species showed increased growth.

Soils

Significant increases of S in foliage, litterfall, forest floor, and soil leachate were associated with increased S deposition from northeastern Minnesota to southwestern Michigan. Levels of N in litter, canopy exchange, and soil O_1 horizons also increased with increasing N deposition along the same gradient. Ratios of S/N in litter increased across the gradient, even though N deposition increased; that is, S in litter increased more than did N. The researchers suggested that this result was due to pollutant deposition. Increasing concentrations of S in the forest floor and soil from west to east have also been reported. In the Ohio Valley, a general increase in total soil S concentrations and soil C concentration has been observed in surveys conducted in 1987-1988. No pattern in soil C concentrations was apparent when county soil survey data of 1960-1970 were reviewed. These results suggest that air pollutant deposition has affected nutrient cycling in northern hardwood forests on a regional basis.

5.1.5 Western Conifers

Chronic symptoms of O_3 damage to ponderosa and Jeffrey pine in the San Bernardino Mountains of southern California have been documented for over two decades. Ozone concentrations have damaged foliage of trees. Concern that foliar injury may lead to changes in forest growth prompted studies to determine whether there have been recent changes in tree growth. No information is available on western conifer mortality.

Growth

In the southern Sierra Nevada (north of the San Bernardino Mountains), sites exposed to O_3 have shown more growth losses relative to increases since 1950 compared with the prior period, although the number of growth increases exceeded the number of decreases. Because these sites also show high levels of foliar injury typical of O_3 exposure, O_3 has been suggested as a cause of the change in relative frequency of growth losses. In southern Arizona, there is evidence of some radial growth reductions in ponderosa pine and Douglas-fir. Researchers have suggested that local pollutant deposition may have affected the climate-tree growth relationship, especially in the south. However, sites located near urban areas (and thus with elevated O_3 exposure) show no consistent growth reductions. Natural competition among trees may be responsible for some of the growth decreases.

In the Front Range of the Colorado Rocky Mountains, the climate-tree growth relationship has shown no widespread change, and changes in growth are not correlated with O_3 levels. Radial growth of Douglas-fir increased in the 1980s and was correlated with increased precipitation. In the Puget Sound area of the Pacific Northwest, radial growth of Douglas-fir increased from 1880 to 1950, remained constant or decreased slightly until 1975, and then increased. The overall increase since 1880 is probably related to a regional temperature increase. No relationships suggest a pollution effect.

Crown Condition

Ozone exposure increases from north to south in the southern Sierra Nevada of California, and crown injury to ponderosa pine occurs over a 250-km corridor in the same direction. The greatest foliar injury is in Sequoia-Kings Canyon National Park. In the Colorado Front Range, no visible injury has been found, although O_3 was at levels associated with chlorosis of needles in the southern Sierra Nevada. This finding suggests that either the duration of exposure to elevated levels of O_3 is less or ponderosa pine in the Front Range are more resistant to O_3 injury than are ponderosa pine in California. In Arizona, O_3 damage has been reported for the Rincon Mountains, where 23% of the trees showed foliar injury.

5.1.6 Wet Deposition

Total wet deposition due to clouds and precipitation in eastern forests was greatest at high elevations with frequent cloudiness. In the West, wet deposition of pollutants was generally less at locations distant from point sources and cities. For areas below cloud base in the Northeast, upper and lower Midwest, and Southeast, precipitation deposition of H^+, SO_4^{2-}, NO_3^-, and NH_4^+ ions was relatively high across each region with no true gradient demonstrated. In states near the edge of these regions (Minnesota, Maine, Arkansas, and Florida), deposition via precipitation was lower than in the remaining portions of the region (Michigan, New York, Illinois, and North Carolina).

In general, the highest deposition of ions by precipitation occurred in the Ohio Valley and northeastern states, and there was relatively low deposition in western forests. Chemical deposition from cloud water in high-elevation areas of the Northeast and Southeast appeared comparable to or greater than deposition from precipitation. A more limited data base indicates that the same may be true in the West. Although direct correlation to changes in forest growth and condition have not been possible, these results illustrate potentially increased risks from pollutants at high-elevation sites.

5.1.7 Ozone

Most O_3 monitoring has been conducted in or near urban areas. Although O_3 from these areas may impact forests because of long-range transport, little is known about actual levels of O_3 in forests. Limited monitoring in forests has been initiated recently in the FRP and other programs. Available data indicate that across the United States, areas in southern and central California, the high desert and intermountain region of the West, and the southcentral East and midwestern states were more likely to be exposed to elevated O_3 levels than were other areas. In Pennsylvania, an O_3 gradient existed during the 1988 and 1989 growing seasons, with the actual levels of O_3 at each monitoring site being higher in 1988 than in 1989. This gradient coincided with the SO_4^{2-} and NO_3^- gradient in Pennsylvania. In the Northeast, high-elevation forests experienced average O_3 concentrations of between 40 to 60 ppb together with some very high peak levels. In the southern Appalachians, O_3 levels increased with the proximity of the site to urban areas, and median O_3 concentrations increased with elevation at remote locations. In midwestern states, forests to the southern portion of the region were exposed to higher O_3 concentrations than were the more northern sections.

In the West, forests in Arizona and New Mexico experienced hourly O_3 concentrations less than 90 ppb, with no distinct spatial pattern in exposure. Forests on western and southern slopes of the San Gabriel and San Bernardino Mountains were exposed to very high hourly O_3 concentrations (median concentrations greater than 100 ppb), and direct injury to vegetation has been documented in the San Gabriel and San Bernardino Mountains of California and the Rincon Mountains in Arizona. Forests in the southern Sierra Nevada were exposed to higher concentrations more frequently than forests to the north. On western slopes of the Cascades, south and southeast of Seattle-Tacoma, forests were exposed to higher concentrations than sites elsewhere in the Puget Sound Basin. Finally, forests in the Front Range of Colorado north of Denver were probably exposed to median O_3 concentrations of approximately 60 ppb, with peaks up to 130 ppb. For other high-elevation Rocky Mountain sites, exposures of 30 ppb were common.

5.2 Conclusions

In this section we discuss the answers to the scientific questions based on the results presented in this document. Changes in forest condition in this report are defined as unexpected or unusual changes in either growth, mortality, or crown condition of forest trees or stands throughout a region of the United States.

5.2.1 Changes in Forest Condition

Individual FRP studies generally evaluated one or more measures of forest condition and accounted for some but not all of the potential sources of variability. Although individual studies have shown local changes in forest condition, only two areas of the United States show regional changes in forest condition. First, the incidence of ponderosa pine foliar injury in the southern Sierra Nevada of California is greater than what would be expected from natural sources of variability. Second, in the Northeast, high-elevation red spruce showed increased mortality in the 1960s. Mortality of high-elevation red spruce in the Northeast has not yet been associated with any natural agents of mortality. The greatest mortality has occurred at or above 1000 m, which may represent an extreme in the range of red spruce. Thus, an environment that may already be considered marginal for red spruce could become intolerable for the species if additional stresses were imposed, such as unfavorable climate and/or the addition of acidic deposition.

5.2.2 Spatial Patterns in Forest Condition and Pollutant Exposure

To determine whether patterns in forest condition are associated with similar patterns in atmospheric deposition, both forest condition and deposition levels must be known at coincident locations. However, deposition monitoring typically has not been conducted at all sites at which forest condition has been assessed. Projects estimate rather than actually monitor pollutant levels at forest plots, and the variability of these estimates is either quite large or unknown. Thus, only forest plots that are a great distance apart are known to have significantly different levels of pollutant deposition. Identification of spatial patterns in deposition over time is further limited because historical data extend back only to the late 1970s, and patterns vary greatly from year to year. Thus, there has been limited success in answering the second scientific question.

Spatial patterns in forest condition are evident for two regions. First, in the southern Sierra Nevada of California, injury to ponderosa pine needles is consistent with symptoms typical of O_3 injury, and the injury is found in areas experiencing elevated levels of O_3. Second, in the Northeast, the percentage of red spruce standing dead basal area increases with increasing elevation in the Adirondack and Green Mountains. It is not known whether mortality rates of red spruce at high elevations are outside the range of natural variability; however, it does appear that rates are above normal expectation and are consistent with increasing levels of wet deposition.

5.3 Recommendations for Future Research

A review of the research on changes in forest condition suggests that many simple correlations that are used to suggest cause-effect relationships often result in oversimplified interpretations of a more complicated system. Retrospective analyses using historical data and long-term monitoring of forests are needed to adequately address whether changes in forest condition are within or outside the range of natural variation. Failure to consider long-term endogenous dynamics of forest stands can lead to spurious correlations and inconclusive studies.

As we have indicated in the document, the research that has been conducted to address the scientific questions has some of these limitations. In this section, we present recommendations for future research on change in forest condition and its relationship with natural factors and atmospheric deposition.

Dendrochronological research can be improved in several ways. Recent interpretations of radial growth data suggest that historical stand development is a major determinant of long-term growth trends, and that stand history has not been adequately considered in relationship with climate and atmospheric deposition. Better growth and yield data and analyses might indicate that many growth reductions are associated with natural factors. Thus, dendrochronological research could be improved by the development of sampling and modeling methods that address both low-frequency (stand dynamics) and high-frequency (climate) variations in tree-ring series.

Work is also needed to determine the magnitude of potential bias in dendrochronological procedures and the implications of extrapolating modeling results to future climate scenarios. Eriksson (1989) recently identified bias in parameter estimates from dendrochronological models that use the sequential process of removing the growth trend and then transforming ring widths to tree-ring indices. The bias results from fitting growth trend equations while ignoring the effects of climate and other endogenous and exogenous variables. Incorrect specification of the growth trend model can lead to bias in coefficients that is then carried through subsequent stages of modeling. In Eriksson's data, the bias was large; however, further work with other data sets has not been completed. The implications of this bias for the autoregressive models of Cook (1985a,b) and Monserud (1986) are unclear. Eriksson states that the sequential method can produce unbiased estimates if subtraction rather than division-based indices and variance stabilizing transformations are used. Further studies are necessary to determine the amount of bias present in commonly used growth-trend functions.

In addition, further investigation of direct estimation is needed. The iterative estimation procedures that transform tree-ring indices are designed to stabilize variances. However, if the variances are not stable to begin with, then applying ordinary least squares at the first step of growth trend removal is inappropriate (Eriksson, 1989). A variance stabilizing transformation, such as a logarithmic transformation, should be applied prior to fitting growth-trend models. This transformation is consistent with the basic model forms often assumed for radial increment.

The selection of climate variables is worthy of attention. In particular, cumulative weather and threshold variables should be developed. Threshold is a critical temperature or moisture deficit that in some manner damages a tree or that causes it to respond to environmental factors in a new or different way. Using within-year variation of an annual ring width density profile instead of using yearly growth may be a way to integrate finer scale threshold events that are likely contributors to growth variation. Cumulative weather is the summation of weather variables over an extended period of time. Quite often the current-year radial increment is preconditioned by past weather variation such that there is a lag between weather and tree growth.

Optimum trend removal from tree-rings in dendrochronology is not straightforward. It might be preferable to focus on trend estimation rather than removal; that is, not only the age trend but also the effects of all exogenous and endogenous disturbances would be tracked. Simultaneous departures from established trends that cannot be associated with known factors might be due to pollution. In addition, the slope of such trends should reveal, for example, whether present reductions in growth rates are greater than previous rates.

As noted by Lucier (1987), neither dendrochronology nor forest mensuration alone has produced an approach that satisfactorily accounts for regional scale changes in forest productivity. Traditional dendrochronological studies do not reveal stand dynamics information over time; only survivors are included (usually dominant or overstory survivors), and these individuals may not reflect stand response. On the other hand, forest growth and yield models usually reflect only stand dynamics while ignoring the influence of climate and other exogenous effects. Thus, each approach quantifies the effects of important sources of variation that the other ignores, and research in combining forest growth and yield information with dendroecology is needed.

In general, populations of trees, forests, or soils have not been formally defined in the FRP. Research findings are difficult to interpret even when populations are clearly defined and corresponding sampling frames are identified. In future programs investigating changes in forest growth, condition and health sampling frames should reflect the fact that tree growth and health are influenced by stand dynamics, species composition, and climate. Progress on identifying sources of variation that influence forest growth and condition could be hastened by sampling stands showing various degrees of decline symptoms. For example, Reams and Huso (in press) hypothesized that birch dieback may have contributed to increased red spruce radial growth from 1935-1955 and a subsequent 1960s growth reduction. Future work should include site visits to check for decomposed birch stumps and interpretation of aerial photographs taken prior to 1940 through the present for permanent plots throughout the Northeast. Such a study could test whether radial growth trends of red spruce are associated with changes in stand structure.

A number of studies in the FRP used point sampling from FIA to estimate stand and tree level attributes. Because coefficient estimates and R^2 values may be over- or underestimated when using large BAF prisms, simulation studies are needed to quantify the level of bias present for different forests and BAFs.

Because changes in growth of southern pines has been restricted to basal area and radial growth, within-tree volume distribution may be different over periods of tree and stand development. Future analyses of tree and stand growth should include changes of volume growth rate.

The eastern hardwoods gradient studies clearly show that only improved identification and characterization of natural factors and supporting data from long-term monitoring of forests will provide the baseline information on forest condition and health that is necessary to quantify the effects of competition, soil moisture availability, climate, and atmospheric deposition at a site. Identification of changes in forest condition also can be improved by the use of formal hypothesis tests. Tests should address whether changes in foliar and crown condition, mortality, and growth rates are associated with each other as well as evaluating what factors are responsible for variability in these measures of forest condition.

Factors influencing the death of trees are poorly understood. Improved identification and measurement of natural factors associated with mortality are necessary. Empirical models of mortality rates also are needed. Such models would help to identify key variables associated with mortality and to determine temporal and spatial trends of change in mortality rates.

Any future programs investigating the effects of air pollution on forest condition must improve pollutant deposition monitoring at forest sites. Analyses of existing monitoring networks should be useful in determining the sampling intensity necessary to capture the spatial and temporal variability of deposition.

Finally, consideration should be given toward integrating existing parallel efforts on soil chemistry. Such efforts include the EPA's Direct/Delayed Response Project and Watershed Manipulation Project, EPRI's Integrated Forest Study, and research by the Canadian Forestry Service. Specifically, criteria for defining and evaluating acid-sensitive soils should be established.

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8 APPENDICES

8.1 Appendix A: Dendrochronological Methods

8.1.1 Introduction

The history of a tree's response to its environment is embodied in its rings. Thus, many studies of the extent and magnitude of changes in forest condition are based on dendrochronological (i.e., dated tree ring) data. Indeed, the temporal nature of tree growth was understood more than 2000 years ago. The association between extreme environmental conditions and abnormal ring development was recognized in 1737, when French naturalists discovered that a frost-damaged ring occurred 29 rings in from the bark on several newly felled trees (Dean, 1978). This feature was subsequently used as a marker for the 1709 ring. (The winter of 1708-1709 had been excessively cold, destroying Charles XII's army in its invasion of Russia).

The ability to crossdate sequences of rings from different trees based on relative ring widths was realized in the 19th century. Nevertheless, only early in the 20th century did dendrochronology become a legitimate field of scientific investigation. The term dendrochronology now typically includes not only the use of tree rings to date particular events, but any endeavor based on accurately dated tree ring sequences, such as dendroclimatology and dendroecology.

8.1.2 Crossdating

The key to any dendrochronological study is accurate crossdating of the tree rings. This can be complicated by false (multiple) and missing rings. Crossdating can be facilitated by graphical techniques and computer analysis (Fritts, 1976). Many recent studies have used a computer program, COFECHA (Holmes, 1983; Holmes et al., 1986), which calculates the correlations between a single series and a master tree series. COFECHA is best used for verification after visual crossdating; it does not replace personal skill and experience. In the following discussion, it is assumed that properly crossdated sequences are available.

8.1.3 Modeling Ring Width Components

Ring growth is the product of many environmental factors, both natural and anthropogenic, acting simultaneously. Natural factors include climate, pests and pathogens and, in some instances, fire. Anthropogenic factors include management practices such as thinning and fertilization, fire, and atmospheric pollution. Thus, to assess the effects of pollutants using tree rings, the pollution signal must be separated from all other factors.

The linear aggregate model (Graybill, 1982; Cook, 1985a) is a conceptual starting point for associating parts of tree-ring series with different environmental factors. If R_t is the observed ring width in year t, then

$$R_t = G_t + C_t + \delta D1_t + \delta D2_t + \delta P_t + E_t$$

where: G_t = the growth trend associated with increasing tree age and size

C_t = climate-related growth variation assumed common to trees in a stand

$D1_t$ = variation due to endogenous disturbances

$D2_t$ = variation due to exogenous disturbances, other than pollution

P_t = variation due to anthropogenic pollutants, affecting a whole stand

E_t = a random component, unique to each tree or radius.

Endogenous disturbances are disturbances affecting a single tree or a subset of trees in a stand. Exogenous disturbances are those affecting radial growth of a whole stand. In the above model, pollution, an exogenous disturbance, is separated from other such disturbances. The δ associated with $D1_t$, $D2_t$, and P_t is a binary indicator of the presence ($\delta = 1$) or absence ($\delta = 0$) of that component in that particular year.

Conceptually, there is little difference between this and Sundberg's (1974) representation:

$$\Delta r(t) = \lambda(t)\, \eta(t)\, [1 - \theta(t)]\, \xi(t)$$

in which $\Delta r(t)$ corresponds to R_t, $\lambda(t)$ is an individual growth level function (cf. G_t), $\eta(t)$ is a climatic factor assumed common to all trees in a region, or having at most a slight smooth spatial trend (cf. C_t). The $[1 - \theta(t)]$ introduces the effect of external agents with $\theta(t) \leq 1$ for t to (cf P_t). Finally, $\xi(t)$ is a random process and it is assumed that log $\{\xi(t)\}$ is a stationary normally distributed process with expectation zero, which is independent for not very closely adjacent trees (cf. E_t).

The linear aggregate model separates endogenous and exogenous disturbances, which Sundberg combines in $\theta(t)$. Another difference between the models is that Sundberg's is multiplicative. The linear additivity of the aggregate model may have been a simplification that was not intended to imply any structural relationship. Indeed, a multiplicative pollution effect might be more realistic than an additive one. The independence of measurements within a tree or stand and among trees or stands also appears unjustified. In addition, models are validated to test for sensitivity over time, but not across different values of the structural components. Care should thus be used if inferences depend on the structure of the components in this type of model.

8.1.4 Standardization

The first step after crossdating is standardization, which is the estimation and removal of the natural growth trend from the raw ring-width sequences for each tree. In the above models, the growth trend (G_t or $\lambda(t)$) reflects, in part, the geometrical constraint of adding a volume of wood to a stem of increasing radius. When this constraint accounts for most of the trend, G_t decays exponentially as a function of time (Cook, 1987). Thus, a typical standardization technique has been to fit a negative exponential function, $\alpha\exp(-\beta t)$ (or sometimes a straight line, α-βt) to the sequence of raw ring widths. A sequence of ring-width indices is then obtained by dividing the raw ring widths by the corresponding fitted trend value. The sequence of indices will have a mean of 1.0 and a fairly stable variance. If standardization is successful, the indices should still contain the signals due to climate and exogenous and endogenous disturbances, as well as random error.

Other functions that have been used or suggested are:

1) the power function, αt^{β} (Kuusela and Kilkki, 1963);
2) the generalized exponential function, αt^{β} exp(-ct) (Warren, 1980);
3) a form based on the Weibull probability density function, $\alpha t\beta$ exp$(-\gamma t\beta)$ (Yang et al., 1978), since any probability distribution function, rescaled, has the properties of a cumulative growth function; and
4) $\alpha(1 + \beta\exp(\gamma t))^{\delta}$ (Van Deusen, 1989).

As noted by Cook (1987), these deterministic models produce monotonic or unimodal curves, which require a simple underlying age trend. Thus, they are best used with open-canopy stands of undisturbed trees or for young trees with strong age trends.

Frequently, the effects of exogenous or endogenous disturbances show up in the raw ring-width sequence as long-term departures from the overall trend. Several ways of removing these along with the natural growth trend have been used:

1) Fitting high-order polynomials (e.g., Jonsson and Matera, 1974): The order of the polynomial is not known in advance and is determined by a stepwise fitting procedure. This method generally has severe end-fitting problems and poor local goodness of fit.

2) Low-pass digital filtering (e.g., Briffa et al., 1983): The digital filter is a centrally weighted moving average of the actual data; the symmetry of the filter thus preserves the original phase information of the sequence. The degree of smoothness achieved depends on the weights selected, for which there is little, if any, theoretical basis.

3) Weighted exponential smoothing (e.g., Barefoot et al., 1974): This is a one-sided causal filter that relies only on the current and prior values of the sequence. There is some phase distortion but, because of the one-sided way that trees grow in time (i.e., they only increase in size), the exponential smoother may be more realistic biologically than the symmetric digital filters. The selection of the weights is again a problem.

4) Cubic smoothing splines (e.g., Cook and Peters, 1981): A succession of cubic polynomial curves are fitted to the data in one pass. This is like a digital filter in which the amount of variance removed at a particular frequency can be specified; more variance is removed at lower frequencies, and less at higher frequencies. In the commonly used computer program ARSTAN (Holmes et al., 1986), the user can specify the 50% cutoff wavelength. The longer the period, the stiffer the spline.

Standardization by fitting exponential functions like these can be described as model-based, although the functions lack flexibility and their appropriateness can be questioned. Once the weights or the stiffness of the spline has been selected, the models are entirely data-based. There may then be no basis for parameter interpretation.

Warren (1980) attempted to introduce some parameter interpretability with the compound exponential model. A general exponential model was fit to the first part of a sequence and extrapolated to the next time period. If the points in this next period were randomly scattered about the extrapolation, the time period was added to the previous, the function was refitted, and the process was repeated. Otherwise, it was assumed that a departure from the trend had occurred, and a new function of the same form was fitted from the estimated point of departure. The method has recently been improved and simplified, and is currently being tested.

Other methods that deserve mention are the corridor method (Shiyatov and Mazepa, 1987), which has not been used in North America, and methods based on differencing (e.g., Van Deusen, 1987a,b; Guiot, 1987). In differencing, the ring-width sequence is considered a random walk with deterministic drift; thus:

$$R_t = R_{t-1} + e_t + m$$

where: e_t = serially uncorrelated random shocks

m = deterministic drift, which imparts a linear trend

The difference $R_t - R_{t-1}$ gives a stationary process, with the magnitude of the drift estimated by the mean of these differences. With tree-ring data, typically logarithms of R_t are differenced. If the negative exponential trend, αt^{β}, were appropriate, the drift in log(Rt) would be $-\beta$. Van Deusen (1987a,b) noted that using logarithms puts the ring widths on a relative scale, so the first difference of the logs is a numerical first derivative of relative growth; that is, a relative growth rate. Therefore, this transformation enhances direct comparisons between trees of different ages and absolute growth rates. The disadvantage of this approach, as noted by Cook (1987), is that there is no explicit estimate of the growth trend, G_t.

Another method recently used in standardization is the Kalman filter. Because it is used for other purposes as well, it is discussed in Section 10.1.9.

There are thus a variety of standardization techniques. They are not equivalent, and little is known about how the method of standardization affects the ultimate inferences. The choice of a method is sometimes governed by the study objectives; for example, a climate signal might be estimated in dendroclimatology, or discarded when studying pollutant effects. Unfortunately, at present no method can be categorized as best in a given context, a characteristic that also applies to all subsequent steps of dendrochronological analysis.

8.1.5 Site Chronologies

The next step is typically to form a site chronology by averaging the indexed sequences of all trees sampled at that site (usually 20 to 30 trees). The site chronology should emphasize the common signal due to climate and exogenous disturbances. However, depending on the method of standardization, exogenous disturbances may have been, at least in part, already removed. In addition, random noise (E_t) and any signals due to endogenous disturbances not accounted for in standardization may have been dampened out. The danger exists that long-term climatic departures from the norm, such as extended drought, might also have been removed by the more flexible standardization procedures. (However, such climatic aberrations can be viewed as exogenous disturbances.)

Rather than calculating a simple average, site chronologies have sometimes been formed using a robust mean, and in particular the biweight mean (Mosteller and Tukey, 1977). The term 'robust' indicates that the influence of outliers is discounted when computing the mean; robust means should thus minimize the effects of non-synchronous endogenous disturbances.

Logically, principal components could also be useful. If the majority of indexed sequences from a site carry a common signal, but a few series have undetected endogenous disturbances, the first principal component should give equal weight to sequences with the common signal, but little or inconsistent weight to the others. If the remaining sequences also carried a common, albeit different, signal, this should show up in the second principal component. Thus, principal components can identify subgroups of trees that respond similarly to each other but differently from other subgroups. Further, by definition, the first principal component should maximize the signal-to-noise ratio. It is unclear whether the principal component has any advantage over the sequence of biweight means, since the latter allocates its weights on a year-by-year basis, while the principal component allocates weights across years. Peters et al. (1981) seem to be the only researchers who have formed site chronologies from principal components. Cook (1988) used principal components to form elevational-stratum chronologies from site chronologies that were based on unusually few (down to three) ring-width sequences.

8.1.6 Time Series Modeling

Standardized tree-ring sequences may not be free from autocorrelation (i.e., year-to-year observations may be directly related), particularly if sustained effects of exogenous and endogenous disturbances have not been removed. Even without such effects there may be persistence; for example, abnormally good growth in one year may carry through to above-average growth the next year. If such autocorrelation within each sequence is high, the site chronology may be more efficiently estimated by prewhitening (i.e., removing the effects of the systematic year-to-year component from the error term, leaving random error) via autoregressive moving average (ARMA) modeling (e.g., Box and Jenkins, 1976).

An autoregressive model for the index values that represents the relationship between adjacent observations may be written as:

$$I_t = \phi_1 I_{[t-1]} + \phi_2 I_{[t-2]} + ... + \phi_p I_{[t-p]} + e_t.$$

where: I_t = response I at time t

ϕ_p = correlation coefficient indicating the magnitude of dependency between adjacent observations

e_t = unsystematic error

The response at time t, I_t, is dependent on the responses of the p previous years; the model is said to be of order p and is written as AR(p). There is thus a carryover effect of previous years plus a random perturbation.

A moving average model accounting for the persistence of random shock adjacent observations may be written as:

$$I_t = e_t - \theta_1 e_{[t-1]} - \theta_2 e_{[t-2]} - \ldots \theta_q e_{[t-q]}.$$

where: I_t = response at time t

e_t = random shock

θ_q = the correlation between adjacent observations

I_t is a weighted combination of the current random shock and those of the immediately preceding years. It is said to be of order q and is written as MA(q). (This model should not be confused with the moving average or running mean filter used to remove certain frequencies for curve smoothing.)

These two models may be combined into an autoregressive moving average model of order p,q; that is, ARMA(p,q). As noted by Monserud (1986), these ARMA models describe stationary time series where the joint probability density between responses at times t and t - s depends only on the interval s and not on time t. The time-dependent growth trend, G_t, should therefore be removed before ARMA modeling.

The order of the process is most commonly determined at the time of model estimation through the Akiake Information Criterion, AIC (Akiake, 1974). Monserud (1986) points out that the percentage of variation explained by the ARMA model should not be viewed as a goodness-of-fit measure, but rather as a measure of how far a tree-ring index series is from being white noise. Monserud performed ARMA modeling for 33 published tree-ring chronologies from western North American conifers, which, if not already standardized, he standardized with either the general exponential function $\alpha \exp(-\beta t)$ or a ninth-order polynomial. The ARMA(1,1) model was judged best in 20 cases and good or adequate in 11 more. The remaining two cases required ARMA(2,2) or ARMA(2,3). The ARMA(1,1) models accounted for from 5% to 45% of the variation of the indexed sequences, with an average of 22%. Monserud concluded that the best competitor to ARMA(1,1) was AR(p) with $2 \le p \le 4$). Cook (1985a), in a study of eastern conifer and hardwood species using the AR(p) model, reported satisfactory results for most cases with $1 \le p \le 3$.

Once the ARMA(p,q) coefficients have been estimated, prewhitening is carried out as:

$$e_t = I_t - \phi_1 I_{[t-1]} - \ldots - \phi_p I_{[t-p]} + \theta_1 e_{[t-1]} + \ldots + \theta_q e_{[t-q]}.$$

The e_t then contains the contributions of C_t, $D1_t$, and E_t. $D2_t$ is assumed to be absent; however, this is not the case unless $D2_t$ has been removed by standardization. As an exogenous effect, it will be contained in the estimate of C_t and will later have to be separated (see Section 8.1.7 below). Clearly, there is a trade-off between how much of the effects of endogenous and exogenous disturbances are removed in standardization, and how much to rely on ARMA modeling, averaging, and later decomposition to remove them.

The next step is the averaging of the e_t as, preferably, a robust mean such as $\overline{e}_t$. Cook (1987) observed that $\overline{e}_t$ may also contain a natural persistence due to climate and tree physiology. A pooled estimate can be obtained directly for a pure AR model, but it is difficult to extend the pooling procedure to the ARMA case. As an alternative, following standardization and averaging as a robust mean, ARMA modeling can be applied to this sequence of means. The viability of this approach depends on sufficient replication to dampen the effects of endogenous disturbance or their removal during standardization.

The sequence should now consist of the common signals of climate and, if not otherwise removed, exogenous disturbances, plus a random error component that should be relatively small. The effect of any endogenous disturbance not removed through standardization plus the random component for an individual tree (i.e., $D1_t + E_t$) can be obtained as the difference between the indexed sequence and this estimate of $C_t + D2_t$. The techniques of intervention detection can then be used to isolate the $D1_t$ (e.g., Reilly, 1984; Chang, 1982, Box and Tiao, 1975). Intervention detection can likewise be used to separate $D2_t$ from C_t.

8.1.7 Climate

To this point the objective has been to estimate the variation in growth due to climate, C_t, free from the natural growth trend and endogenous or exogenous disturbances, including atmospheric pollution. If this component can be accurately described by meteorological measurements, it may be possible, after allowing for the effects of age and disturbances, to predict ring widths of trees thought to be affected by pollutants. Consistent overestimation (or underestimation) could be evidence of a negative (or positive) effect of pollution and provide a measure of the effect's magnitude.

The conventional method of relating the estimated C_t to meteorological variables has been multiple linear regression. To detect or quantify a pollution effect, the relationship is estimated from the period deemed to be relatively pollution free, sometimes assumed to be all years before 1960. Following Fritts (Fritts et al., 1971,1979; Fritts, 1976; Ashby and Fritts, 1972), the meteorological variables are typically monthly mean temperatures and total precipitations for both the growth year and the preceding year. Commonly, regressions have had approximately 28 potential climatic predictors. In addition, mean indices from the previous one to three years have sometimes been included, primarily when persistence has not otherwise been accommodated.

To avoid multicollinearity problems (i.e., when predictors are not independent), the principal components of the weather variables have sometimes replaced actual monthly values. The number of potential predictors has sometimes then been reduced by omitting components that account for negligible variability amongst the raw predictors. However, the principal components may be no more than an arithmetic artifact; there is no a priori guarantee that any can be readily interpreted as, for example, winter rainfall or spring temperature.

A forward-selection stepwise regression is then conducted. In several instances, inclusion of a potential predictor has been based on the sequential F statistic being greater than 1.0. This is a liberal criterion, since in the stepwise procedure, under the null hypothesis of no association, the expected F value would be greater than d/(d-2), where d is the number of degrees of freedom in the denominator of the F ratio. On completion, the multiple coefficient of determination, R^2, is usually computed and its significance judged by reference to standard tables. The estimated regression is conventionally known as the response function.

Claims of significance based on this procedure are, unfortunately, not justified. The probability values for both F and R^2 statistics obtained from standard tables are not applicable for steered regression, of which forward selection is a particular case (Rechner and Pun, 1980). The critical values of a test of the null hypothesis at any specified level exceed those in standard tables, usually substantially. Reliance on conventionally computed probabilities or tabled critical values has thus sometimes led to undue confidence in the regressions so derived for predicting response to climate. It no doubt accounts for Cropper's (1982) finding more significant weather variables as predictors of a random number sequence than of tree ring data.

Another drawback of this approach is that, from a physiological point of view, an additive linear model of monthly temperature and precipitation is at best a crude predictor. For example, the effects of temperature and moisture may not be additive; for a given temperature, if moisture is adequate, why should additional moisture generate any response? Conversely, elevated temperatures may have beneficial or detrimental results depending on moisture availability. This problem exists whether monthly values or principal components are used. The Palmer Drought Severity Index (PDSI), a measure integrating moisture availability and temperature, seems beneficial (Palmer, 1965). Seasonal values, with seasons determined from biological considerations rather than by calendar months, also hold potential; with fewer potential predictors, spurious significance may be reduced.

Because of these difficulties, instead of modeling the estimated C_t signal based on meteorological records, it may be better to look for associations between departures from the norm of both C_t and weather (cf. Cook et al., 1987). Threshhold values based on long-term records may be used to determine departure from the norm; for example, temperature or precipitation might not be expected to exceed extremes more than one time out of 20.

8.1.8 Verification

Predictive equations are usually not accepted without verification. Verification is commonly accomplished by splitting the pre-pollution period into two data sets. Regression parameters are first estimated with a calibration set, and the resulting equation is applied to the predictors of a verification set. The predicted and observed values are then compared. Goodness-of-fit criteria are calculated, such as bias, the coefficient of determination (R^2), and the reduction of error (RE) statistic (Lorenz, 1959; Kutzbach and Guetter, 1980; Fritts, 1976). If the goodness-of-fit criteria are met, the two data sets are combined, and the parameter estimates from the overall regression are then used for prediction and comparison in the pollution period. As an additional check, the procedure is sometimes repeated, reversing the calibration and verification sets.

The goodness-of-fit statistics should not be attributed properties that they do not possess. Indeed, the properties of the RE statistic have not been formally derived and its use has therefore been heuristic. For example, the RE statistic is given by:

$$RE = 1 - \Sigma(y_i - \hat{y}_i)^2 / \Sigma(y_i - \bar{y}_c)^2$$

where: y_i = actual response values

$\hat{y}_i$ = estimated response values

$\bar{y}_c$ = mean value from the calibration set

Summation is over the verification set. Positive values of RE are assumed to indicate good predictability. They may arise because $\Sigma(y_i - \hat{y}_i)^2$ is small, as would be hoped, and/or because $\Sigma(y_i - \bar{y}_c)^2$ is large. The latter may result if the verification period is different from the calibration period, particularly if the verification period is relatively short (as might be the case with a pollution period).

The standard statistical approach of comparing the regressions fitted to each period against a single regression for the combined periods does not appear to have been used in this context by any researcher.

8.1.9 The Kalman Filter

The Kalman filter, initially developed by Kalman (1960) for system and control theory, is the most recent tool used in tree-ring series analysis (e.g., Visser, 1986; Visser and Molenaar, 1986; Van Deusen, 1987a,b). The fundamental difference between the Kalman filter and techniques discussed above is that the system parameters are regarded as time-dependent. There are distinctions in terminology between Van Deusen (1987a,b) and Cook's (1987) application of the Kalman filter; Van Deusen's is used here.

The relationship between the vector of observations at time t, Y_t, and the parameters, α_t is called the measurement or observation equation and may be written in generic form as follows. :

$$Y_t = F_t \alpha_t + v_t.$$

where: F_t = a fixed matrix

v_t = a vector of residuals with expectation zero and variance matrix V_t.

alpha sub t] may be thought of as state parameters that evolve over time according to a first-order Markov process; thus:

$$\alpha_t = G_t \alpha_{t-1} + w_t$$

where: G_t = a fixed matrix

w_t = a vector of residuals with expectation zero and variance matrix W_t.

The latter equation is referred to as the transition or system equation. The means of solving the resulting system of equations will not be discussed here. The variance matrices V_t and W_t are assumed known in advance; in practice, they are selected by trial and error or estimated by maximizing the logarithm of the likelihood function. The likelihood function is specified completely in terms of the prediction errors and their covariance matrices, which are the natural outputs of the Kalman filter solution; the only assumption is that w_t and v_t are normally and independently distributed.

In their initial application of the Kalman filter, Visser and Molenaar (1986) took Y_t as the sequence of standardized tree-ring indices and for F_t they substituted z_t, a vector containing all selected meteorological variables and lagged indices at year t. (All explanatory variables were transformed to have mean zero and unit variance.) Their transition equation was a simple random walk. Their model was thus:

$$Y_t = z_t \alpha_t + v_t$$

$$\alpha_t = a_{t-1} + w_t.$$

The α_t estimates, along with their confidence intervals, may be plotted against time to determine whether the response versus climate relationship is stable or has shifted (as might occur if pollution started to have an influence).

Cook (1987) reports that H. Visser and J. Molenaar (personal communication) have applied this approach to standardization. The negative exponential is taken as the basic trend model, with logarithms of ring widths then taken to achieve linearity, and thus expressed as the sum of the stochastic trend plus noise. Both the slope and intercept are allowed to vary through time as random walks. Thus:

$$R_t = b_{0,t} + e_t$$

$$b_{0,t} = b_{0,t-1} + b_{1,t-1} + \eta_t$$

$$b_{1,t} = b^{1,t-1} + \zeta_t.$$

Van Deusen (1987a,b; 1988) gives two examples of Kalman filter applications to tree ring data, one of which will be outlined here. The starting point is a standard sigmoid model for the diameter of a tree at age A, D(A) = b(1 - exp(-kA)), from which is derived:

$$Y_t = \Delta\log[R(t)] = C_t + D1_t + D2_t + e_t.$$

The objective is to form simultaneously a single chronology and its climate-based prediction. The Kalman filter formulation is then the observation equation:

$$Y_t = \alpha_{1,t} + v_t$$

and the transition equations are:

$$\alpha_{1,t} = c_t \, \alpha_{2,t-1} + w_{1,t}$$

$$\alpha_{2,t} = \alpha_{2,t-1} + w_{2,t}$$

where: c_t = the climate variable

$\alpha_{1,t}$ = the value of the chronology at time t

$\alpha_{2,t}$ = the climate effect at time t.

In more recent work, Van Deusen (1989) suggested a Kalman filter formulation designed to allow all the traditional steps in dendroecological modeling to be handled simultaneously. Application of the Kalman filter is still a new approach that has been used in different ways for different parts of the process. It remains to be determined whether its use to incorporate all steps simultaneously is viable or even desirable.

8.1.10 Discussion

A variety of strategies have been used to analyze dendrochronological data and, for each strategy, a variety of procedures. Removing the age-dependent growth trend in standardization seems universal, but trends due to endogenous and/or exogenous disturbances may or may not be removed. If the focus is the effect of climate, then removal of these disturbances is essential (although extreme climatic phenomena may be regarded as exogenous disturbances, in which case it should not be removed). However, if the focus is on pollution (also an exogenous disturbance), it is not desirable to remove its signal without identifying it (as, perhaps, an exogenous disturbance with no other explanation). Recent work by Eriksson (1989) has shown that the sequence of age-trend removal and standardization introduces bias. She suggested techniques of alleviating or estimating any bias that may be introduced.

If the focus is climate, prewhitening may seem desirable, although it may do little if it follows a flexible method of standardization. On the other hand, Van Deusen (1989) argues that autocorrelation in the residuals of the climate model, not in the standardized data, is the appropriate concern for valid climate modeling. Because prewhitening removes autocorrelation from the response variable, rather than from the model residuals, he states that it is not justified and that non-independence of residuals should be handled in model estimation.

Forming a site chronology by averaging individual (prewhitened) standardized series may also be questioned. Again, if the focus is climatic relationships, averaging should highlight the signals of interest. Since meteorological records are rarely available for the exact sites where trees were sampled, more meaningful results have been obtained by forming a regional chronology from several site chronologies and then relating this to weather data from several meteorological stations in the region. To determine whether a particular stand is influenced by pollution, this technique is less appropriate. Because of genetic variation and competition, it may be desirable to explore the response range within a stand and estimate, for example, the proportion of trees exhibiting decline. In sampling trees to date historical events or for climatic reconstruction, purposive selection of sensitive trees is justified. To make inferences about the growth trends and condition of a whole forest, such subjective sampling is less appropriate.

Finally, one might ask whether developing response functions, while relevant, is necessary to identify and quantify pollution effects. If pollution has an effect, it should show up as a departure from an established trend as would any exogenous disturbance. Unless there is a contemporaneous departure in climate, the magnitude of the pollutant effect could be estimated as the difference between the extrapolation of the previously established trend and the actual present trend. This assumes that there are no confounding exogenous disturbances, such as insect infestations. Unfortunately, such disturbances are not always recorded. In addition, there may be synergistic effects of a pollutant with one or more other stressors, even though it may be difficult to identify a response to any one factor by itself.

Although most dendrochronological studies have used tree-ring widths, more information is potentially available. Tree rings are identifiable by the transition from higher density latewood (produced in the later part of the growth season) to lower density earlywood (produced in the first part of the next growth season). X-ray densitometry can generate density profiles that include several measures: ring width; widths of the early- and latewood components and, thus, the proportion of latewood; maximum and average latewood density; minimum and average earlywood density; and overall density.

Using density as a measure of tree response, Parker et al. (1974) reported a decrease in maximum density in trees subjected to a point source pollutant relative to trees in a control area. Conkey (1987) examined maximum wood densities of high-elevation red spruce in Maine since 1962. A flattening of the high-frequency variation in density, coupled with a lower mean value, led Conkey to suggest that wood density may be an early signal of growth decline. There are few facilities for obtaining wood density profiles, however, and it is a costly process, so few dendrochronological studies of forest condition include this measure. The forestor's traditional concern with volume or basal area production, sometimes at the expense of wood quality (of which wood density is perhaps the most important component) also contributes to the lack of research that includes wood density.

Even more detailed information is available through image analysis of the anatomical structure of tree rings (e.g., Yanosky and Robinove, 1986). Different information may be obtained through elemental analysis of tree rings (e.g., Hall, 1987; McClenahen et al., 1989). Indeed, tree-ring widths are only the end products of a process; any association between them and environmental factors is correlative at best. To properly understand the process, it may be necessary to generate more basic information; for example, measures such as photosynthesis rate may be useful.

8.2 Appendix B: Quality Assurance

The Quality Assurance (QA) Project was initiated in late 1985 to fulfill US-EPA QA requirements for research funded by the FRP. Although QA had been used in EPA environmental monitoring, QA principles had never been formally adapted to a diverse forestry research program prior to the FRP. The objective of EPA's QA programs is to ensure that data quality is known and documented.

There are two types of QA activities: internal and external. Internal QA consists of project-level QA implementation, including methods documentation and collection of quality control (QC) data. External QA includes the review of internal project QA and, for some types of research, the comparison of QA activities across projects. In the FRP, the QA staff provides external QA and assists in the development of internal QA project activities.

As QA was implemented, FRP investigators were expected to:

1) develop a QA Project Plan to accompany the research plan;

2) participate in audits conducted by the QA staff;

3) collect, use, and archive QC data; and

4) participate in relevant training sessions and sample exchanges with other researchers.

Several projects that were in progress prior to full implementation of QA did not fully participate in QA/QC activities. Table 14 indicates the participation of FRP projects cited in this report. Most of the non-FRP research discussed did not have any formal external QA.

8.2.1 QA Project Plan

A QA Project Plan (QAPP) is a detailed description of research procedures and methods as well as a documentation of all internal QA/QC activities. A QAPP is written by the principal investigators for each project, and it contains descriptions of:

1) the project and its relationship to other FRP projects;

2) experimental design, sampling procedures, and statistical methods and analysis;

3) project management and personnel;

4) research facilities and equipment;

5) measurement and analytical procedures;

6) data quality objectives (DQOs) and data quality assessment procedures;

7) sample and data custody and archive procedures;

8) QC activities;

9) data validation and analysis;

10) mechanisms for implementing changes in research or QA

The QAPP is reviewed and approved by the QA staff. QAPP development is an iterative process, with new information being added as requested and as research methods are changed or developed. At the beginning of the FRP QA project, methods manuals were developed to provide guidance and standardize procedures by describing basic methods and presenting DQOs for relevant variables (Zedaker and Nicholas, 1988; Robarge and Fernandez, 1987; Evans and Dougherty, 1986). The level of detailed documentation required in a QAPP was new to most researchers and is essential to ensure data quality.

Table 14. Summary of QA/QC activities for FRP projects cited. Projects are arranged by forest cover type and data source.

Project No.	Investigator	Research Type	QA Plan	Audits	QC Data
		Northern Red Spruce			
Use of Existing Databases					
VS09	Van Duesen	Dendrochronology	N[1]	N	N
SF28	Solomon	Modeling, Growth	Y	N	N
Fixed Plots					
SF05	Conkey	Dendrochronology	Y	Y	Y
SF05	Hornig	Insect/Pathogen Survey	Y	Y	Y
SF08	A. Johnson, Silver, Peart	Soil/Insect/Pathogen Survey, Dendrochronology	Y	Y	Y
SF	Friedland	Insect Survey	Y	N	N
VS14	Parker Miller-Wee	Insect/Pathogen/ Damage Survey	Y	Y	Y
FIA					
VS06	Hornbeck	Dendrochronology	N	Y	Y
Other					
AE01	Mohnen	Deposition Monitor.	Y	Y	Y
SF15	Wargo	Insect/Pathogen/ Damage Survey	Y	Y	N
		Southern Red Spruce			
Review of Existing Literature					
SF18	Wentworth	Literature Review	N	N	N
Use of Existing Databases					
VS09	Van Duesen	Dendrochronology	N	N	N
Use of Existing Databases					
SF02	Bruck, Hartman	Insect/Pathogen Survey	Y	Y	N
SF25	Zedaker	Growth, Damage Survey	Y	Y	Y
Use of Existing Databases					
AE01	Mohnen	Deposition Monitor.	Y	Y	Y
SF26	Dull	Remote Sensing, Damage Survey	N	N	N
SC10	Christensen	Modeling, Growth	N	Y	N

Table 14. (continued)

Project No.	Investigator	Research Type	QA Plan	Audits	QC Data
		Western Conifers			
Use of Existing Databases					
WC36	Böhm	Modeling	Y	N	N
Fixed Plots					
WC24	Graybill	Dendrochronology	Y	Y	Y
WC25	Brubaker	Dendrochronology	Y	Y	Y
WC26	Peterson	Dendrochronology	Y	Y	Y
WC29	Peterson	Dendrochronology	Y	Y	Y
Other					
WC03	Larson	Deposition Monitor.	Y	Y	Y
WC04	Miller	Deposition Monitor.	Y	Y	Y
		Eastern Hardwoods			
Review of Existing Literature					
EH02	Millers	Literature Review (+E. white pine)	Y	N	N
Fixed Plots					
EH03	Witter	Deposition Monitoring, Biomass, Nutrient Cycling	Y	Y	Y
EH05	Loucks	Deposition Monitoring, Damage Survey, Dendrochronology	Y	Y	Y
EIA					
VS06	Hornbeck	Dendrochronology	N	Y	N
VS10	Ohmann	Dendrochronology, Nutrient Cycling	N	Y	Y
VS11	Brooks	Growth, Soil Survey Dendrochronology	N	Y	N

[1] Y, QC data was collected and is available in QA files, Annual/Final reports, and/or at the project site. N, data are not available.

8.2.2 QA Audits

QA audits are on-site reviews of research facilities by a QA specialist. Also evaluated are sampling and experimental methods and QC practices. Audits are conducted to identify concerns early, to provide feedback to researchers, and to document QC practices. The QAPP serves as the basis for the audit, and the plan is checked against actual methodology. An audit report, summarizing the review and recommendations, is submitted to both the researcher and the FRP QA Coordinator at the Corvallis EPA Environmental Research Laboratory. Significant problems were brought to the attention of the FRP Research Cooperative Manager.

Technical systems audit review procedures are used in sample collection and analysis, measurement of variables, and data collection and analysis. Areas checked during audits include: project organization and personnel; equipment and facilities; calibration and maintenance records; personnel performance and support systems; QC activities; sample collection, handling, custody, and storage; data verification, analysis, and synthesis; and items identified during a previous audit as requiring corrective action.

The FRP projects in Table 14 that were not audited generally were either completed or in the final stages of data analysis before the QA project was implemented. Since then, the goal has been to audit each project early in the research schedule, and then once each year during the most active periods of research.

8.2.3 Collection of QC Data

QC data are collected to track, document, and assess data quality. In the QAPP, DQOs and acceptable range limits were stated for the variables being measured or estimated. DQOs were either taken from the QA Methods Manuals or established by researchers as the degree of tolerable variability (precision) and accuracy. Researchers are encouraged to collect QC data on all variables being measured, to compare QC data to DQOs, and to use the comparative results as feedback for improving the research.

QC data can be used to quantify accuracy, precision, and comparability. Accuracy, estimated by comparison with standards or reference materials, is a measure of the closeness of individual measurements to a true value. Precision, estimated by comparing replicates or by evaluating repeated analyses of a certified reference material, is a measure of agreement among individual measurements of the same variable. Comparability, estimated by comparing values obtained by different methods or recorded by different individuals or crews, is a measure of agreement and an indicator of bias. QC data collected on some variables has facilitated estimation of previously unknown sources of error.

The type of QC data collected depends on the variable being measured. Methods for collecting QC data are standard for laboratory research, moderately well-developed for dendrochronological work, and in early stages of development for field studies. Table 15 describes QC data collected for different FRP research types. Analytical laboratory QC data may include analyses of replicates, blinds, and reference standards, spike recovery efficiency of analytical equipment, checks on distilled/deionized water quality, balance calibrations, and purity checks on reagents. Dendrochronological QC data includes remeasurement of ring widths and results from COFECHA, a software program that verifies visual crossdating of tree rings. QC data of field projects includes remeasurement data of quantitative variables, test results of field crews, and calibration of field equipment.

The QA staff has developed programs for collection of QC data across projects. Joint training sessions of field survey crews were coordinated to assess comparability of field data collection procedures and to minimize group differences. Interlaboratory exchanges of soil, water, and foliar material are under way to assess comparability of analytical laboratories in the FRP.

Table 15. Description of QA project plan, audits, and collection of QC data for different research types.

Research Type	QA Requirement
Dendrochronology	QA Project Plan (QAPP) QAPP may document project specific methods of sample collection and handling of data. Manuals and standardized methods for core collection and preparation, and measuring ring widths may be cited.
	Audits Audits include review of ring measuring techniques, instrumentation, computer facilities, sample and data handling and storage.
	QC Data Well-developed (but varied) techniques for crossdating, recording missing or irregular rings. Software developed (e.g., COFECHA) for estimating measurement error and technician bias in ring width data.
Inventory/Growth Estimates/Yield	QA Project Plan QAPP may cite QA methods manual or FIA inventory procedures (may vary by region)
	Audits Reveiw of data collection procedure can be accomplished during aduit. Most FIA crews are also audited by field crew supervisors.
	QC Data Includes formal crew training, calibration of field equipment and remeasurement of field data.
Literature Review	No applicable QA/QC principles. When possible, data quality in reviewed literature is assessed.
Modeling	QA Project Plan Some quidelines for documentation have been developed by QA staff.
	Audits Few applicable QA/QC principles.
	QC Data Some QA/QC activities have been developed for assessing quality of existing databases.

Table 15. (continued).

Research Type	QA Requirement
Remote Sensing (Damage Survey)	QA Project Plan Some QA/QC principles; generally involve data management issues such as data validation and verification.
	Audits General review of data management procedures.
	QC Data some procedures involving checks on measurement and area delineation on maps.
Deposition Monitoring; Nutrient Cycling; Soil, Precipitation, and Throughfall Chemistry	
	QA Project Plan QAPP may document sample collection and storage, data management. Existence of standardized and well-documented SOPs and QA practices for most lab analyses. Manual developed for the FRP is a standard reference, others are also cited.
	Audits Highly developed techniques for review of calibration records, instrument maintenance records, personnel responsibility.
	QC Data Highly developed QA principles and activities; QC data may include control charts; precision and accuracy of analysis based on analyzing replicates and reference standards, spike recovery efficiency of analytical equipment; balance calibration, purity checks on reagents, estimates of technician bias.
Survey (Soil/Damage/Insect/Disease)	QA Project Plan Field methods manuals standardize collection of field mensurational, damage, and symptomatology data. QAPP documents plot establishment and training to maximize comparability among crews, and cite methods manual.
	Audits Well-developed techniques for review of survey methodology. Also QA staff review and participate in field crew training sessions.
	QC Data Well-developed QA/QC principles include joint training of field crews, calibration of field equipment, remeasurements of quantitative data, joint assessment of crown class estimates.

8.2.4 Conclusions

The QA project has improved the quality of data collected in the FRP in four ways:

1) The requirement for project-specific QAPPs has resulted in documentation of methods and protocols, thus increasing the potential use and value of FRP data.

2) The use of QA Methods Manuals has standardized research activities and increased comparability and understanding of data.

3) Audits have identified problems in research and led to corrective action and problem resolution.

4) The collection and reporting of QC data have provided the means for quantitative evaluation of data quality.

8.3 Appendix C: Abbreviations

AARI	Average Annual Radial Increment
AIC	Akiake Information Criterion
AIRS	Aerometric Information Retrieval System
ARMA	Autoregressive Moving Average
AVI	Annual volume increment
BAF	Basal Area Factor
dbh	Diameter at breast height
DISP	Drought Index for Southern Pines
DQO	Data Quality Objective
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
FIA	Forest Inventory and Analysis
FRG	Federal Republic of Germany
FRP	Forest Response Program
GIS	Geographic Information System
IFS	Integrated Forestry Study
MCCP	Mountain Cloud Chemistry Project
MPO	Major Program Output
NADP/NTN	National Acid Deposition Program/National Trends Network
NAPAP	National Acid Precipitation Assessment Program
NF	National Forest
NP	National Park
NPS	National Park Service
p	probability value
PDSI	Palmer Drought Severity Index
QA/QC	Quality Assurance/Quality Control
QAPP	Quality Assurance Project Plan
S&I	Synthesis and Integration
SAROAD	Storage and Retrieval of Aerometric Data
SoCAB	South Coast Air Basin
SOP	Standard Operating Procedure
TVA	Tennessee Valley Authority
USDA	United States Department of Agriculture
USFS	United States Forest Service

8.4 Appendix D: Scientific Names of Trees

American beech	*Fagus grandifolia* Ehrh.
Balsam fir	*Abies balsamea* (L.) Mill.
Bigtooth aspen	*Populus grandidentata* Michx.
Black cherry	*Prunus serotina* Ehrh.
Black oak	*Quercus velutina* Lam.
Chestnut oak	*Quercus prinus* L.
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco
Eastern white pine	*Pinus strobus* L.
Engelmann spruce	*Picea engelmannii* Parry
European beech	*Fagus sylvatica* L.
Fire cherry	*Prunus pennsylvanica* L.
Fraser fir	*Abies fraseri* Poir.
Green ash	*Fraxinus pennsylvanica* March.
Hophornbeam	*Ostrya virginiana* Mill.
Jack pine	*Pinus banksiana* Lamb.
Jeffrey pine	*Pinus jeffreyi* Grev. and Balf.
Limber pine	*Pinus flexilis* James
Loblolly pine	*Pinus taeda* L.
Lodgepole pine	*Pinus contorta* Dougl.
Northern red oak	*Quercus rubra* L.
Norway spruce	*Picea abies* (L.) Karst
Paper birch	*Betula papyrifera* Marsh.
Pinyon pine	*Pinus edulis* Engelm.
Ponderosa pine	*Pinus ponderosa* Laws.
Quaking aspen	*Populus tremuloides* Michx.
Red maple	*Acer rubrum* L.
Red pine	*Pinus resinosa* Ait.
Red spruce	*Picea rubens* Sarg.
Shagbark hickory	*Carya ovata* (Mill.) K. Koch
Shortleaf pine	*Pinus echinata* Mill.
Silver fir, European	*Abies alba* Mill.
Southwestern white pine	*Pinus albicaulis* Engelm.
Sugar maple	*Acer saccharum* Marsh.
White ash	*Fraxinus americana* L.
White fir	*Abies concolor* (Gord. and Glend.) Lindl.
White oak	*Quercus alba* L.
White spruce	*Picea glauca* (Moench) Voss
Yellow birch	*Betula alleghaniensis* Britton
Yellow-poplar	*Lireodendron tulipifera* L.